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INTRODUCTION

This report represents an effort to summarize current knowledge regarding the potential effects of toxicants on organisms that inhabit the San Francisco Bay and Delta. It encompasses studies conducted in the Estuary, as well as describes potential effects, based on knowledge gained from other systems. It also makes recommendations designed to focus further research in areas for which the data are not sufficient to support conclusions. The report addresses toxicants in the Bay and Delta and generally considers the upstream limits to be Sacramento on the Sacramento River and Vernalis on the San Joaquin River. However, potential impacts upstream of these points were addressed if they pertained to species or processes thought to be important to the Estuary. In general, only water-borne contaminants and impacts were considered although issues related to sediment were described if they were of particular importance or related to specific species of interest.

The San Francisco Bay and Delta is located at the confluence of the Sacramento and San Joaquin Rivers. [add brief description of system]

Other investigators have suggested that contaminants may have contributed to the decline of Bay and Delta resources. [summarize past compilations, including AHI]

This review is divided into sections designed to present the material in a manageable manner. Evidence of toxicity in samples collected from ambient waters associated with the Bay and Delta is presented first, since this is potentially the strongest evidence that contaminants may be present at levels that result in adverse effects. This presentation is followed by a summary of impacts from non-point sources, which include agricultural and urban stormwater inputs into the Bay-Delta system. In some cases, such as the Colusa Basin Drain, agricultural sources constitute large identifiable inputs into the system. Conversely, numerous small agricultural drains and return pumps contribute to waterways throughout the system. Urban stormwater inputs are similar in that they often comprise most of the high flows of local creeks and storm channels but the inputs are diffuse and not readily treatable.

Next, toxicity and contributions from point source dischargers are reviewed. These dischargers include publically owned sewage treatment works (POTWs), industrial dischargers, and oil refineries. Any toxicity associated with these discharges is compared to the potential for the receiving water to dilute toxic components to levels below those associated with adverse effects.

The section that describes species effects deals with data directly related to the Bay/Delta system, as well as data on the same species found in different systems. This is because the data that demonstrate effects of contaminants on organisms that inhabit the Bay and Delta are limited and it is helpful to have information gathered in other studies to use as a basis for generating information that could be applied locally. These data may help identify key species and parameters that can be incorporated into monitoring programs.

The Discussion section attempts to pull all of the information together, provide an indication of the effects of contaminants on the system, and point out data gaps that should be filled to provide a solid basis for generating decisions that deal with maintaining a productive estuary complex. Key points from the Discussion are presented as Conclusions and Recommendations.

METHODS

Information was gathered from a variety of sources. A comprehensive literature review was undertaken to identify recent publications in the peer-reviewed literature on areas of interest. Species of interest were identified through a document that described the biotic resources of the Bay and Delta (). Species included in the search are listed in Appendix A. Citations found in the literature search were retrieved directly form library stacks, inter-library loans, or through literature retrieval services. Each article was then reviewed and summarized and included in this document as appropriate.

State agencies and university researchers were also contacted to determine whether pertinent unpublished reports or data were available. If available, these reports or data were reviewed, summarized and, where appropriate, included in this document. The different agencies and researchers contacted are summarized in Appendix B.

To further ensure as much coverage as possible, draft copies of this document were passed back through the various agencies and researchers for review. Thus, they had an additional opportunity to comment as well as point out studies that may have been missed in the initial data screening process.

Ambient Toxicity

Comparatively few studies have investigated toxicity in ambient waters in the Bay/Delta system. In 1986, toxicity was noted in the Sacramento River at Sacramento using *Ceriodaphnia dubia* as the test organism (Foe and Connor 1991). In 1987, follow-up work suggested that much of the toxicity was related to seasonal discharges associated with rice culture and toxicity was found in the Sacramento River approximately 2 miles downstream of Colusa Basin Drain, which enters the River at Knight's Landing. Elevated mortality was also seen with fathead minnows exposed to water collected from this site. Similar results were obtained in 1988; however, toxicity was found both above and below Colusa Basin Drain which made it difficult to separate the sources of toxicity. In 1989, monitoring with *C. dubia* at the end of May showed elevated mortalities at all sites on the river down to Rio Vista, 75 miles downstream of Colusa Basin Drain. One week later, elevated mortalities appeared to be limited to the stretch 30 miles downstream of the Drain (Foe and Connor 1991). Follow-up Toxicity Identification Evaluations (TIEs) conducted by EPA indicated that carbofuran and methyl parathion, two pesticides applied to rice, were responsible for toxicity to *C. dubia* in 1988 (Norberg-King *et al.* 1991). TIEs conducted on samples collected from CBD in 1989 suggested that carbofuran and methyl parathion were responsible for toxicity to *C. dubia* (Norberg-King *et al.* 1989).

Bailey reported reduced survival of striped bass larvae exposed to one sample collected from the Sacramento River at Rio Vista on 15 May 1988 (Appendix B in Foe and Connor 1991). In 1989, two samples collected from the Sacramento River at Walnut Grove produced nearly complete mortality of larval striped bass within 96 hr. These samples were collected at the end of May and early June during the rice discharge season.

Finlayson *et al.* (1993) #35 reported on the toxicity of samples collected from the Sacramento River at Rio Vista during rice season 1990. Three of the 30 samples significantly ($p<0.05$) reduced the survival of *N. mercedis* within 96 hr. These samples were also tested with striped bass larvae, but high and variable control mortalities made it problematic to assess the extent of toxicity to this species. The authors noted that the cause of the striped bass

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decline could be due to direct effects on striped bass and/or to their food organisms.

In late winter 1991-1992, eleven sites on waterways tributary to San Francisco Bay/Delta were monitored for toxicity associated with orchard run-off (Foe and Sheipline 1993). The sites were divided between the Sacramento and San Joaquin Basins. Six sites represented watersheds that ranged between 10,000 and 130,000 acres, with at least 10 percent of this total in orchards. The remaining five sites represented larger water bodies, including the Sacramento River, Feather River, Mokelumne River, Old River, and the San Joaquin River. Intermittent toxicity was observed in the smaller drainages during dry weather. However, all of these drainages exhibited toxicity during the period of precipitation that occurred between 4 and 20 February 1992. Of the larger water bodies, only the Mokelumne River exhibited toxicity during the dry period. However, once runoff occurred, samples from the Feather, Old and San Joaquin rivers also exhibited toxicity. Toxicity persisted in the San Joaquin River for some time following the storm. A total of 25 samples exhibited toxicity; in 22 cases, the organophosphorous pesticides diazinon and/or methidathion were present at concentrations high enough to produce acute mortality. Concurrent chemical monitoring data suggested that the San Joaquin river was acutely toxic for at least 8 days (12-19 February) and that this water reached as far north as Empire Tract and Venice Island before being diluted by flows from the Sacramento and Mokelumne rivers. These investigators concluded that the toxicity was largely due to the presence of pre-emergent pesticides, primarily diazinon, that had been applied to the orchards and were transported off-site in run-off from rain.

Toxicity associated with run-off from sites in the Delta that grew alfalfa was evaluated during March and April 1992 (Foe and Sheipline 1993). Intermittent toxicity was seen, particularly in Ulatis Creek, Bishop Tract Main Drain, and Paradise Cut. Analytical results indicated that both carbofuran and diazinon were present at potentially lethal concentrations in some of the samples. The authors felt that the data were representative of a "dry year" because no precipitation related run-off occurred during the study period.

Toxicity in samples collected primarily from Hog, Sycamore and Beaver Sloughs was evaluated with *C. dubia* in May and June 1994 (DiGiorgio *et al.* 1994). Samples were collected weekly from different sites, but only one of the 60 samples collected exhibited toxicity to *C. dubia*.

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This may have been a consequence of the time of sample collection; very little pesticide applications were occurring in these watersheds during the study period and no rainfall events occurred.

Delta sites were monitored twice monthly between May 1993 and May 1994, alternating between sites on the Sacramento and San Joaquin sides, using short-term chronic tests with fathead minnows, *C. dubia,* and *Selenastrum capricornutum* (Deanovic *et al.* in prep.). Sampling sites included major river and channels, back sloughs, and island drains. The results, based on a preliminary analyses of the data, are summarized below:

Sacramento side	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY
No. tested	15	11	12	15	12	12	- -	12	- -	- -	13	12	12
No. toxic	7	5	6	7	8	4	- -	8	- -	- -	4	1	4
San Joaquin side													
No. tested	12	12	12	12	12	12	12	- -	12	- -	- -	12	13
No. toxic	5	9	5	3	1	5	2	- -	0	- -	- -	3	4

During most of the year, adverse effects were observed in approximately 33 to 75 percent of the samples with one, or more, of the test species. Months in which ≤ 10 percent of the samples toxic were April (Sacramento River side of the Delta) and September and January (San Joaquin side of the Delta). In most cases, toxicity was associated with the smaller creeks and sloughs and the Island drains. However, larger waterways were also frequently affected. In ten testing events conducted between May 1993 and May 1994, toxicity was found in samples collected in the Sacramento River (6 events), San Joaquin River (3 events at Vernalis and 5 events at Antioch), Old River (7 events), Middle River (1 event), and the Mokelumne River (5 events). The Port of Stockton site, which was considered to be dominated by urban inputs, exhibited toxicity in 4 events. Samples collected from the Delta-Mendota Canal also exhibited toxicity in 4 sampling events. These numbers would appear to be of concern; not only were the smaller creeks and sloughs affected, but also large masses of water moving through the Delta appeared to be contaminated as well.

Species sensitivity varied. In most cases, only one of the test species responded to a particular sample, implying that different toxicants were responsible. Based on a preliminary analysis, a total of 42 samples exhibited toxicity to *C. dubia,* 36 samples were toxic to fathead minnows, and 28 samples were toxic to *S. capricornutum.* Only 7 samples exhibited toxicity to both fathead minnows and *C. dubia* and 6 samples exhibited toxicity to *C. dubia* and *S. capricornutum.* None of the samples exhibited toxicity to both the alga and fathead minnows.

Toxicity was also monitored during four rainfall periods (Bailey *et al.* in prep.). In the first period, 6-hr composite samples were collected over a two-day period at Green's Landing on the Sacramento River. One of the six samples collected exhibited toxicity to all three of the test species, including increased mortality in *C. dubia* and fathead minnows.

In the second period, samples were collected daily between 23 and 28 January 1994. 100 percent mortality was seen with *C. dubia* in samples collected from the San Joaquin River at Vernalis from 24 through 27 January. A follow-up TIE suggested that the mortalities were due to metabolically activated organophosphorous pesticides. The sample collected at Vernalis also reduced algal growth to approximately half of that seen on the four previous sampling days. No adverse effects were observed with *C. dubia* or the alga exposed to samples collected from the Sacramento River at Green's Landing during this same sampling event. However, growth was significantly reduced in fathead minnow larvae exposed to samples collected at Green's Landing 24 and 25 January.

A third series of tests evaluated toxicity at the same sites during another precipitation event (6-13 February 1994). Toxicity to *C. dubia* was apparent in samples collected 8-11 February at Vernalis. None of the samples collected at Vernalis exhibited toxicity to fathead minnows, but the sample collected 9 February reduced algal cell numbers. One sample collected from the Sacramento River (11 February) exhibited toxicity to *C. dubia* and one sample also exhibited toxicity to fathead minnows (10 February). None of the samples collected from the Sacramento River exhibited toxicity to the alga.

Additional samples were collected from Vernalis between 17 and 23 February 1994. None of these exhibited toxicity to any of the three test species.

Samples were collected in the Sacramento river at Garcia Bend between December 1990 and November 1991 and tested for toxicity with *C. dubia* and fathead minnows (AQUA-Science 1993). Of the nine samples collected, 6 significantly increased mortality in the minnows tested. No adverse effects were noted with *C. dubia*. Between February 1992 and November 1992, the sampling site was changed to Freeport. Of 6 samples tested with fathead minnows, one produced elevated mortality and another reduced growth. Two of the samples also reduced the survival or reproduction of *C. dubia*. Three additional collections were made at Freeport between November 1993 and March 1994 and ambient toxicity evaluated with the 7-day fathead minnow test (AQUA-Science 1993 and 1994). No toxicity was observed in the sample collected in November, but survival and growth were adversely affected in the sample collected in February 1994. A follow-up sample collected in March also exhibited reduced survival.

EPA (1991) published a technical support document for water-quality based toxicity control. In this document, the Agency provides rationale for the development and application of water quality standards based on toxicity which are designed to protect aquatic communities. For acute toxicity, a maximum 1-hour exposure duration is permitted. For chronic or sublethal effects, the maximum allowable duration is 4 days. For both criteria, exceedences are permitted only once in 3 years, which gives the natural community a three year recovery period. The Agency also provides documentation that toxicity measured in samples with the standard EPA 3-species test[1], which was used to generate most of the data described above, should provide a level of detection within one order of magnitude of the most sensitive species found in the receiving water. Thus, while the absence of toxicity in the 3-species test does not necessarily mean that all organisms will be protected, the presence of toxicity strongly suggests that there will be effects on the receiving water biota.

How EPA's criteria relate to the Delta has not been established. However, the Agency's conclusions are based on a number of studies conducted by the EPA and independent investigators that relate toxicity to observed instream effects in both fresh and saltwater. These comparisons, which include data from over 200 sites, indicated that adverse effects could be predicted on the basis of toxicity in approximately 90 percent of the cases (EPA 1991). This

[1]Test species include *Ceriodaphnia dubia* (invertebrate), *Pimephales promelas* (fish), and *Selenastrum capricornutum* (alga). See EPA (1991b) for test procedures.

high percentage of prediction, coupled with the diversity of sampling sites, suggests that this relationship is robust and should also apply in the Delta.

Clearly, toxicity occurs on a frequent basis in water samples collected from the Delta and its tributaries. In some cases, toxicity appears to be associated with rainfall events, while in other cases, toxicity can be related to local inputs. Toxicity may also be related to specific cropping practices, such as rice production, although recent data suggest that more restrictive pesticide use requirements have reduced toxicity associated with this particular crop (Bailey *et al.* 1994b). In any case, the frequency of toxicity suggests that EPA's guidance for maintaining water quality is being exceeded in the Delta much more frequently than the one event in 3 year interval estimated to be necessary to allow for recovery of an impacted system.

Non Point-Source Discharges

Agricultural Inputs – Agricultural inputs may include suspended sediment, various fertilizers, and pesticides. High concentrations of ammonia may be associated with discharges from dairies and feedlots. In general, pesticides appear to be the major source of toxicity in agricultural waters dischared to the waters that enter the Delta. Determination of adverse effects associated with agricultural inputs into Estuary waters has been largely ignored until fairly recently. This has been reflected in lack of analytical monitoring for associated pesticides in Bay and Delta waters, as well as in lack of toxicity testing of these waters. However, as a result of studies that suggested that pesticides were adversely affecting water quality in the Estuary and its tributaries (see, for example, Bailey *et al.* 1994; Finlayson *et al.* 1993; Foe and Sheipline 1993; Foe and Connor 1991; Norberg-King *et al.* 1991; Foe and Connor 1989), California Department of Fish & Game initiated the process of producing water quality criteria for pesticides of interest. Criteria have been developed for molinate and thiobencarb (Harrington 1990), carbofuran (Menconi and Gray 1992), methyl parathion (Menconi and Harrington 1992b), and chlorpyrifos (Menconi and Paul 1994). A draft document for diazinon has also been completed (Menconi and Cox 1994). Values of 0.5 and 0.08 µg/L were recommended for carbofuran and methyl parathion, respectively, to protect aquatic life. An interim value of 0.02 µg/L was recommended for chlorpyrifos and diazinon concentrations protective of acute and chronic toxicity were 0.08 and 0.04 µg/L, respectively.

Rice is single largest use of irrigation water in Sacramento valley. Rice return flows, which can comprise up to 33% of Sacramento River flow, are discharged along a 90 km stretch of river between Colusa and Verona at the mouth of the Feather river. Discharge from rice culture may account for as much as 25 – 33 percent of the total flow of the Sacramento River (Cornacchia et al. 1984). Significant inputs to the River include Colusa Basin Drain, Butte Slough and Sacramento Slough. Input from Colusa Basin Drain alone can account for approximately 25 percent of the flow of the Sacramento River in some years. Most of the discharge enters the Sacramento River upstream of Sacramento. However, in years of high River flow, discharge from Colusa Basin Drain may be diverted into the Yolo Bypass and enter the River via Prospect Slough, downstream of Sacramento. Because of the comparatively large contribution of rice return flows to the overall flow of the Sacramento River, the capacity for dilution of incoming toxicants is relatively low.

Monitoring of the Delta for the rice herbicides molinate and thiobencarb was not initiated until 1985, even though large fish kills associated with agricultural drainage from rice culture had been observed several years earlier (State Water Resources Control Board 1990). Measured concentrations of molinate and thiobencarb in 1985 suggested that concentrations toxic to *Neomysis mercedis* were approached in the upper Delta (Bailey 1993). In subsequent years, more restrictive pesticide management practices (increased on-field holding times) reduced concentrations of these pesticides in the Delta to levels below those associated with toxicity, although molinate and thiobencarb have been detected in the Delta as recently as 1993 (K. Kuivila, USGS, personal communication). In the years between 1982 and 1985, extrapolation from measured concentrations in Colusa Basin Drain suggests that the toxic threshold for these pesticides could have been exceeded by a factor of nearly six for *N. mercedis* (Bailey 1993). Analytical data for earlier years are not available but comparisons of application rates and river flows suggests that discharge concentrations back to 1978 and 1981 were at least as high as those present between 1982 and 1985 for molinate and thiobencarb, respectively (see Table 1, Appendix A).

Other pesticides applied to rice have also been associated with toxicity. Bailey *et al.* (1994)

showed that five pesticides out of approximately 20 applied to rice for at least five years between 1970 and 1989 were negatively correlated with striped bass recruitment and, in fact, could explain nearly 90 percent of the variation in recruitment during this period. The pesticides were bufencarb (now discontinued), carbofuran, methyl parathion, molinate, carbaryl, and MCPA. With the exception of molinate and MCPA, these pesticides are cholinesterase inhibitors and would be expected to exert additive toxicity. Since most of these pesticides were applied in conjunction with each other, there could have been additive effects or effects on food organisms in addition to direct effects on striped bass early life stages. One pesticide in particular, bufencarb, exhibited extremely high toxicity to striped bass embryos and larvae, with acute LC50s of approximately 0.1 μg/L. At this level of toxicity, a daily input of 2400 g would be sufficient to poison the Sacramento River at a flow rate of 10,000 cfs. This would equate to approximately 150 lbs. over a 30-day period; 150 lbs. is only 0.2 percent of the average amount applied annually between 1973 and 1981.

Application of carbofuran and methyl parathion to rice increased in 1980-1982 as bufencarb was being phased out. Applications to rice and Sacramento River flows between 1970 and 1988 are shown in Table 2, Appendix A. Although the amounts applied were appreciable, monitoring of carbofuran concentrations was not initiated until 1987. In that year, concentrations as high as 2.1 μg/L were detected in the Sacramento River during rice season (Menconi and Gray 1992). This value is clearly in excess of the water quality criteria of 0.5 μg/L recommended by DF&G (Menconi and Gray 1992). Based on application and flow rates, it is likely that 2.1 μg/L was exceeded annually between 1980 and 1987 and the criterion of 0.5 μg/L was exceeded back through 1977. More restrictive use requirements reduced concentrations in the River to ≤ 0.5 μg/L by 1991 (Menconi and Gray 1992). Monitoring in the River and Delta between 1990 and 1992 by USGS suggests that carbofuran discharged from rice still reaches the Delta in trace concentrations (Kuivila *et al.* 1992). These data also suggested that Delta inputs of this pesticide, primarily from alfalfa, may also be significant. The half-life of carbofuran in natural water was 3 weeks (Sharom et al in #33).

Intermittent monitoring of methyl parathion concentrations in the Colusa Basin Drain and Sacramento River has occurred since 1980. In 1988 concentrations as high as 0.32 μg/L were detected in the Sacramento River. This value exceeds the recommended water quality criteria of

0.08 μg/L for this pesticide and also exceeds the acute LC50 values for *Daphnia magna* and *Neomysis mercedis* (Menconi and Harrington 1992b). Based on applications and flow rates (Table 2), River concentrations could have exceeded 0.32 μg/L in four of the nine years between 1980 and 1988. Using similar reasoning, the water quality criterion would have been exceeded in 15 of the 19 years between 1970 and 1988 (all of the years after 1976). Menconi and Harrington (1992b) also concluded that levels of this pesticide could have exceeded the criterion during the early 1980s. Their reasoning was based on a 25 percent contribution from Colusa basin Drain to the Sacramento River and measured concentrations of 3.7 μg/L in Colusa Basin Drain. This could have resulted in concentrations in the Delta of up to 0.94 μg/L, a level considerably higher than the acute LC50 for *N. mercedis*. More recent monitoring data for 1990 sugggest that more restrictive management practices have decreased River concentrations of methyl parathion to ≤ 0.1μg/L (Menconi and Harrington 1992b).

The fragile relationship between management practices and off-site movement of pesticides from rice culture is shown in the following table (data from DPR 1994).

Pesticides Transported in the Sacramento River Past Sacramento (kg)

Year	Molinate	Thiobencarb
1988	3194	68.1
1989	1984	11.4
1990	3204	51.2
1991	99	0
1992	57	0
1993	2007	0

For comparison, an estimated 18,465 kg molinate was transported in 1982.

In 1991 and 1992, loadings of these pesticides in the Sacramento River decreased by over an order of magnitude from levels seen in previous years due to the new management plans. However, in 1993, emergency releases from pesticide-treated fields prior to completion of the on-field holding time requirements resulted in the highest pesticide loadings to the River in five years, in spite of the fact that these releases were associated with < 3 percent of the total acreage treated. In fact, the loadings to the River were even higher than shown in the table since flows from Colusa Basin Drain, the largest single source of rice pesticides to the Sacramento River, were diverted into the River downstream of DPR's monitoring point for rice pesticides

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(DPR 1994). Based on concentration and flow data in Colusa Basin Drain given in DPR (1994), a conservative estimate of additional loadings into the Sacramento River would be 1550 and 80 kg of molinate and thiobencarb, respectively. Regional Water Quality Control Board staff compared the toxicity of samples collected from discharges from fields undergoing emergencey releases and from fields that had reached the required holding times (Schnagl and Wyels 1993). Water samples from fields that had complied with the required holding times were not toxic to *C. dubia* while nine of ten tailwater samples collected from fields undergoing emergency releases were acutely toxic to *C. dubia.*

Pesticides entering the Sacramento-San Joaquin system are also associated with other types of agriculture, discharges from municipal sewage treatment plants and storm water run-off. In two years of sampling the Delta, the three pesticides responsible for most of the observed toxicity were carbofuran, chlorpyrifos, and diazinon (Bailey *et al.,* unpublished data). These pesticides occurred in locally high concentrations in smaller waterbodies or in large quantities of water moving through the Delta during major storm events. Selected samples that exhibited toxicity and the associated pesticide concentrations are shown in the following table. All of the samples were acutely toxic to *C. dubia* and Toxicity Identification Evaluations were performed to determine the cause of toxicity.

Sample Site	Date	Pesticide	Concentration (µg/L)
French Camp Slough	3-23-94	chlorpyrifos	1.2
Paradise Cut	4-27-94	carbofuran	9.4
Paradise Cut	4-30-94	carbofuran	7.0
Paradise Cut	7-12-94	chlorpyrifos	0.6

Foe and Sheipline (1993) monitored watersheds in the Sacramento-San Joaquin Basin associated with orchards for toxicity during the dormant spray season between 13 January and 27 February. A total of 11 sites were monitored over 7 sampling events during this period. At least one sample collected from 9 of the 11 sites produced ≥ 50 percent mortality in *C. dubia.* Two, or more, samples collected from five of the sites produced total mortality in *C. dubia.* Of the 25 samples that exhibited significant mortality, diazinon was present in 22 samples at concentrations that exceeded the acute water quality criterion proposed by CDF&G (Menconi and Cox 1994). The median value was approximately 7 times the criterion, but concentrations as

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high as 6.8 µg/L (over 80 times the criterion) were reached. 21 of the samples contained concentrations that have been shown to cause acute mortality in *C. dubia* and six of the samples contained diazinon at concentrations lethal to *N. mercedis.*

An example of movement off-site into local receiving waters during irrigation and precipitation events is shown in a CDF&A study on diazinon applied in the lower American River watershed (Segawa and Powell 1989). In a three-year emergency program designed to eradicate the Japanese beetle, it was found that most of the diazinon was confined to upper layers of soil and thatch, but that significant off-site movement in irrigation water and stormwater run-off occurred. Concentrations as high as 73 µg/L occurred in creeks receiving irrigation run-off and a concentration of 82 µg/L were recorded in local streams following rainfall events. Rainfall events as low as 0.4-0.6 cm were sufficient to move significant quantities off-site. Mass discharge rates of 7.8 gm/hr were recorded during irrigation and as high as 24 gm per hr during rainfall events. During rainfall events, discharge rates as high as 5100 µg/sec were measured. Following a rainfall event in Nov 1993, diazinon concentrations at nine sampling sites ranged between 0.4 and 44 µg/L. These values are from 1 to 110 times the acute LC50 for *C. dubia* and all exceed the DF&G draft criteria for diazinon. The median concentration was 2.9 µg/L, 7.25 times the LC50 for *C. dubia* and 2.4 times the LC50 for *N. mercedis.* In spring 1984, concentrations in the streams ranged between 0.2 and 82 µg/L, with a median of 4.9 µg/L. Measurements in Arcade Creek in fall 1984 during irrigation were between 0.7 and 11 µg/L, with a median of 6.4 µg/L. During rainfall events, concentrations reached 21 µg/L, 52 times the LC50 for *C. dubia.* Although less pesticide was applied in 1985 and 1986, concentrations still reached 2 and 27 µg/L in Arcade Creek in 1985 during irrigation and run-off periods, respectively. In 1986, rainfall events produced in-stream concentrations of up to 4.2 µg/L.

Using the fall (Aug - Oct) treatments as an example, Segawa and Powell (1989) estimated that an average of approximately 50 gm/day was leaving the treatment areas via waterways, with peaks of up to 200 gm/day. Assuming a uniform discharge rate, 50 gm/day would have been sufficient to contaminate a flow rate of 51 cfs at the approximate *C. dubia* LC50 of 0.4 µg/L. During precipitation events, up to 24 gm/hr was estimated to leave the treatment areas via waterways (18 gm/hr in one creek!). Using similar reasoning, this amount would render a flow rate of 600 cfs acutely toxic to *C. dubia* or cause a flow of 3000 cfs to exceed the DF&G acute criterion for diazinon.

Kuivila (1994) tracked diazinon concentrations in the Sacramento River at Sacramento and in the San Joaquin River at Vernalis prior to and during rainfall events in early February 1993. On the Sacramento River, it was found that pulses of diazinon moved past the City of Sacramento 1-3 days after each rainfall event. Each pulse lasted 4-5 days. Diazinon peaks in the River reached 0.4 µg/L, compared with pre-event concentrations of 0.03-0.05 µg/L.

Kuivila (1993) tracked the first Sacramento River diazinon pulse downstream into the Estuary. Her data suggest that diazinon concentrations peaked at Rio Vista and Chipps Island approximately 1- and 3 days after the pulse was recorded at Sacramento. Peak concentrations at Rio Vista and Chipps Island were 0.3 and 0.2 µg/L, respectively. It took an additional three days for the peak (now approximately 0.1 µg/L) to reach Martinez. The decreasing concentrations were due to tidally induced mixing. In addition, the peaks broadened as the pulse moved downstream. At Sacramento, concentrations exceeded 0.1 µg/L for five days, compared with 8 days at Chipps Island. All of these concentrations exceeded the acute diazinon criterion proposed by DF&G.

In contrast to the Sacramento River, diazinon concentrations at Vernalis responded in bimodal peaks following the first rainfall event, suggesting upstream as well as local sources of diazinon (Kuivila 1994). Background concentrations of diazinon at Vernalis prior to and between storm events were approximately 0.1 µg/L. In the first event (1.8 inches of rain), concentrations rose above 0.3 µg/L, with peaks of 0.8 and 1.1 µg/L, for 8 days. The two subsequent events were much smaller, 0.7 and 0.6 inches, and resulted in 48- and 24-hr peaks of 0.3 and 0.2 µg/L, respectively. Similar data were also found in the San Joaquin River at Stockton during the same events. All of these concentrations exceeded the draft acute criterion for diazinon proposed by DF&G.

In contrast to the pulses observed on the San Joaquin River, diazinon concentrations at sites on the Old and Middle Rivers increased steadily from 0.04 to 0.15 µg/L during this same period

(Kuivila 1993). These latter sites do not have a pronounced downstream flow gradient and are heavily influenced by tidal movements. All of these measurements were ≥ the chronic criterion proposed by DF&G and some also exceeded the acute criterion.

In terms of toxicity, 100 percent mortality was observed with *C. dubia* exposed to samples collected daily from the San Joaquin River at Vernalis for 12 days following the first event (Kuivila 1993). Diazinon concentrations in these samples were ≥ 0.15 μg/L. Other pesticides, including chlorpyrifos, methidathion, and carbaryl may also have contributed to toxicity in these samples. No toxicity was observed in samples that contained ≤ 0.08 μg/L diazinon.

The USGS also monitored pesticides in the Sacramento River at Sacramento and in the San Joaquin River at Vernalis between 1 December 1993 and 28 February 1994 (MacCoy 1994). A total of 124 samples were collected, divided almost equally between the Sacramento River and San Joaquin Rivers. Diazinon was not detected in 26 percent of the samples collected from the Sacramento River. 34 percent of the samples exceeded the draft CDF&G 4-day average criterion (chronic) of 0.04 μg/L diazinon, and 13 percent of the samples exceeded the 1-hr criterion (acute) of 0.08 μg/L. The results were very similar for the San Joaquin River; diazinon concentrations exceeded the draft chronic and acute water quality criteria in 44 and 16 percent of the samples, respectively, and was not detected in 33 percent of the samples. Chlorpyrifos was not detected (0.025 μg/L = D.L.) in any of the samples, but other pesticides were found, most notably simazine (to a maximum value of 1.7 μg/L) and methidathion (up to 1 μg/L). Generally, elevated concentrations of pesticides appeared in fairly distinct pulses that lasted between 3 and 10 days.

Interestingly, the amount of diazinon transported in the Sacramento River during these February events was much higher than in the San Joaquin River. Even though maximum concentrations were approximately 2.5 times higher at Vernalis than in the Sacramento River, flows in the Sacramento were 10 to 15 times greater than in the San Joaquin River (Kuivila 1994). Depending on the interactions between pesticide applications, rainfall events, and flow, much higher concentrations could be carried in the Sacramento River into the Delta. Since average flows in the Sacramento River in February between 1987 and 1991 were 12,800 cfs (CV=24.5%), compared with the 40,000-60,000 cfs that occurred during this study, it would appear that the potential exists for substantially higher concentrations to occur in the Sacramento River than measured in February of 1993.

This analysis is supported by data collected in February 1994 by DPR (V. Connor, RWQCB, personal communication). In this study, diazinon concentrations as high as 0.7 µg/L, nearly nine times the proposed acute water quality criterion, were measured in the Sacramento River following a rainfall event of 1.6 inches in four days. River flow rate varied between 12,000 and 30,000 cfs during this period.

Menconi and Cox (1994) presented monitoring data for diazinon collected from 47 sites in the Sacramento-San Joaquin System between March 1991 and February 1993. A total of 340 samples were collected. Measured concentrations ranged between 0.01 and 36.8 µg/L diazinon. A total of 104 (30.6 %) samples exhibited diazinon concentrations that were less than the draft chronic water quality chronic crtierion. 170 of the samples (50 %) exceeded the acute criterion and the remainder exceeded the chronic criterion. All of the samples collected at Freeport/Rio Vista (n=4), Vernalis (n=6), Chipps Island/Martinez (n=3) exceeded the acute criterion of 0.08 µg/L.

In their water quality document for chlorpyrifos, Menconi and Paul (1994) presented monitoring data collected from sites in the San Joaquing system between March 1991 and February 1993. Of the 25 sites sampled, chlorpyrifos concentrations exceeded the criterion at 17 sites. Of the sites samples at least five times, only the Stanislaus River consistently exhibited chlorpyrifos concentrations less than the criterion. Six of the 45 samples collected from the San Joaquin River exceeded the LC50 for *C. dubia.* Concentrations as high as 0.35 µg/L were recorded from the San Joaquin River; this value exceeds the LC50s for *C. dubia* and *N. mercedis* by factors of 5 and 4, respectively.

Taylor et al (1991) #253 evaluated the effect of 3,4-dichloroaniline (DCA) (a hydrolysis product of several herbicides), atrazine, copper, and lindane on *Chironomus riparius* and *Gammarus pulex.* Relative toxicity varied with chemical and exposure period up to 10 days exposure. 10 day LC50 estimates for *C. riparius* were 4.2,18.9, 0.2, and 0.013 mg/L, respectively. For *G. pulex,* the LC50s were 5.0, 4.4, 0.033, and 0.007 mg/L. Other data of interest that were presented included a *C. tentans* LC50 of 0.72 mg/L atrazine, and 0.16 mg/L DCA for *D. magna.*

Gilliom and Clifton (1990) reported total DDT concentrations of 0.01-0.08 µg/L in water samples collected from the San Joaquin River at Vernalis. These values exceeded the EPA 24-hr water quality criterion by factors of 10-80. Based on sampling conducted in 1985, Gilliom and Clifton concluded that concentrations of organochlorine pesticides in bed sediments of the San Joaquin River were among the highest measured in major rivers in the United States.

Saiki et al (1993) # ? evaluated boron, molybdenum and selenium in aquatic organisms in the lower San Joaquin drainage. Concentrations of boron and selenium were elevated in reaches that received tile drainage from irrigated agriculture. Boron and molybdenum were not biomagnified in food chain. Selenium appeared to be biomagnified. Selenium concentrations in some areas in fish reached 23 µgSe/gm body wt. (dry weight), twice as high as needed to elicit reproductive effects. Boron levels were also somewhat elevated. Chinook salmon and striped bass fingerlings accumulated up to 200 µg/g B after 28 days of exposure to tilewater and also exhibited poor survival. Threshold Se concentrations associated with reproductive failure in fish include: 2-5 µg/L in water, 4 µg/g in sediment, 5 µg/g in food, and 12 µg/g in whole fish. Whole fish concentrations as low as 3-8 µg/g reduced growth and survival in juvenile chinook salmon. Authors concluded that any increase in tile drainage to the San Joaquin River will further increase adverse effects on fish.

Saiki et al. (1992) #222 evaluated selenium and other elements in freshwater fish from diferent sites in the San Joaquin valley. Arsenic, Hg, and selenium were elevated in fish from one or more sites but no evidence for accumulation of Cr was obtained. Tissue concentrations of Se were high enough in some cases to adversely affect survival growth and reproduction. The distribution of elevated Se concentrations coincided with inputs of tile drainage to the San Joaquin River, primarily through Salt and Mud Sloughs. The data suggested that Se concentrations peaked in 1984 and have declined slightly since. Changes in agricultural practices could reverse this decline. Fish from the site just downstream of the confluence of the Stanislaus River exhibited body burdens of Se that ranged between 1.3 and 1.8 µg/g dry wt. The values were similar for mosquito fish, bluegill, carp and largemouth bass. Much higher levels were seen in Salt and Mud Sloughs (up to 11 µg/g), and even higher concentrations were found in the mid-1980s (up to 23 µg/g). The data suggest that the problem currently is relatively local, although more may have been exported in the past. Chinook salmon and striped bass exposed to tile water accumulated Se and showed reduced growth. This could be a problem if Se concentrations increase as a function of increased tailwater discharges into the river.

Horne (1991) #246 evaluated the effects of permenant flooding of Se-contaminated lands in the San Joaquin drainage on Se concentrations. Permenant flooding rendered Se unavailable as an insoluble fraction in anoxic sediments, thus making it biologically immobile. Se concentrations in *Chara* (macro alga), an herbivorous chironomid, and the predaceous damselfly were monitored following flooding. Rapid initial declines in Se concentration were followed by long term decrease. Decreases noted in water column and in organisms over the 2.3 year study period. Between 85 and 93 percent of the initial Se was lost; no difference was seen betweeen herbivore and predator. Concentrations in *Chara* dropped to 3-4 ppm and to 14-15 ppm in insects.

In the fall 1983 treatment, there were nine confirmed bird kills (no numbers given) associated with the diazinon treatment (Segawa and Powell 1989).

Urban Stormwater Inputs – In the spring of 1993, Hansen and Associates (1994) investigated toxicity associated with stormwater in the San Lorenzo Creek watershed which enters San Francisco Bay just north of Hayward. Samples were collected from 3-5 sites in the watershed following three precipitation events. Diazinon concentrations at the sites ranged between 0.74 and 2.9 µg/L for samples collected 16 March, between 0.82 and 2.9 µg/L for samples collected 17-21 March, and between 0.08 and 0.46 µg/L in samples collected 7 April. With the exception of the sample that contained 0.08 µg/L, all of the measured concentrations exceeded values associated with acute toxicity in *C. dubia* and two of the values exceeded the acute LC50 for *N. mercedis*. All of the values exceeded the proposed CDF&G acute and chronic water quality criteria for diazinon.

In the fall and winter of 1994 – 1995, stormwater samples were collected from creeks and sumps discharging into the lower Sacramento and American Rivers, and also into the lower San Joaquin River (Bailey *et al.* unpublished data). Samples collected from Arcade, Elder, and Strong Ranch Creeks following separate precipitation events all exhibited acute toxicity to *C. dubia*. In all cases, toxicity was removed by treatment with piperonyl butoxide, a biochemical that inhibits the toxicity of metabolically activated organophosphorous pesticides. Diazinon was found at acutely toxic concentrations in all three creeks and chlorpyrifos was found at acutely toxic concentrations at 1 of the 3 sites. In one of the two sumps, diazinon and chlorpyrifos were both present at acutely toxic concentrations, but metals also contributed to toxicity. In the remaining sump, toxicity was driven by high zinc concentrations. Algal cell numbers were also reduced when exposed to the stormwater samples. Stormwater samples collected during the 1993 – 1994 season also produced mortality in exposed fathead minnow larvae.

Sites that drained into the San Joaquin River in the vicinity of Stockton also exhibited acute toxicity to *C. dubia* and to the algae over multiple testing events (Bailey *et al.* unpublished data). These sites included Calaveras River, Smith Canal and Mosher Creek. A fish kill was also observed in the Calaveras River during the first precipitation event of the season.

Point-Source Discharges

Chapman et al (1994) #22 evaluated the toxicity of refinery waters to a variety of aquatic organisms. There was a wide range of sensitivity observed but some species of fish and invertebrates were adversely affected at ≤ 10 percent effluent. The authors also noted eroded fins. Weiss et al (in #22) found effects on winter flounder, striped bass, and mummichog at 10 percent effluent (growth and development). With fathead minnows, DeGreave et al (in #22) found NOECs of 0.5 and 21.6 percent.

Pesticides have also been associated with acute toxicity in municipal sewage treatment plants (Amato *et al.* 1992). Locally, metabolically activated organophosphorous pesticides were found to be responsible for acute toxicity to *C. dubia* in 10 of 14 toxic samples collected from a 150 mgd treatment plant that discharges into Suisun Bay (AQUA-Science 1992). A Toxicity

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Identification Evaluation demonstrated that diazinon was the primary toxic constituent and chlorpyrifos also contributed to toxicity.

A study conducted on five POTWs that contribute a total daily flow of approximately 235 mgd to the Estuary suggests that the potential for toxicity due to pesticides may be widespread (Miller *et al.* 1994). Based on weekly samples collected for a 6-week period, all of the POTWs contained measurable levels of chlorpyrifos and diazinon in their influent and effluent streams. Four of the five plants contained at least one sample that had diazinon concentrations in excess of levels associated with acute *C. dubia* LC50s. Samples from three of the five plants contained chlorpyrifos at acutely toxic concentrations. The plants varied markedly in the removal efficiencies associated with the pesticides, particularly with respect to chlorpyrifos, which suggests that toxic concentrations may be treatable within the context of plant operation.

Ankley et al. (1990) #40 demonstrated that surfactants may be responsible for toxicity in municipal effluents.

Smith and Bailey (1990) #141 investigated attraction and avoidance of anadromous fish to refinery (San Francisco Bay) and municipal effluent (Russian River). The test species included steelhead, striped bass, and Chinook salmon. Both attraction and avoidance responses to refinery effluent were noted, depending on concentration. Responses were found at dilutions as low as 1000:1. At concentrations of ≤ 100:1, attraction was noted for all three species. Results appeared to be validated by field observations; for striped bass, juvenile fish found in the slough that received the discharge exhibited eroded fins, suggesting adverse effects coupled with lack of avoidance response. Adult salmon were also frequently found in the slough that received the discharge. Steelhead were the only species tested with domestic effluent; in contrast to the attractant response to refinery effluent, strong avoidance to the POTW discharge was noted. Consequently, operation of POTW treatment plant was modified to permit intermittent discharge during steelhead migration.

Insert to point source dischargers pg.18.

An Effluent Characterization Program was initiated with dischargers into San Francisco Bay By the Regional Water Quality Control Board. Under this program, dischargers were required to regularly monitor the acute and chronic toxicity of their effluent for approximately 12 months. Sampling intervals were generall monthly, with intensive weekly testing conducted for short periods of time. Both marine and freshwater species were used, the selections depended on the sensitivity found during a screening phase. Data for these studies are summarized in the attached tables, which indicate the test species, the number of samples tested and the number that produced toxicity [only one table is attached; additional data reduction is ongoing]. Evidence of acute and chronic toxicity was found for most of the dischargers evaluated. [this discussion will be more quantitative as the analysis is completed]

Sediment Toxicity

Swartz et al (1994) #23 compared sediment toxicity, contamination and amphipod abundance at a site in San Francisco Bay contaminated with DDT and dieldrin. The Lauritzen and Santa Fe Channels and part of Richmond Inner Harbor have been designated as a Superfund Site. Property adjacent to the sites was used to formulate DDT and dieldrin from 1945 to 1966. In some areas, largely removed in 1990, banks along the Lauritzen Channel contained virtually 100 percent DDT. Sediment concentrations were highest in Lauritzen Channel, decreasing through the Santa Fe Channel to Richmond Inner Harbor. Sediment samples were evaluated for toxicity with *Eohaustorius estuarius*, analyzed for concentrations of contaminants, and compared with respect to amphipod abundance in the field. Except for one site in the Santa Fe Channel with high PAH levels, concentrations of PAHs, PCBs, and metals were not high enough to cause toxicity. Threshold sediment toxicity occurred at 300 µg DDT/g organic carbon (OC) for *E. estuarius* and at 100 µg DDT/gOC for the naturally occurring populations of amphipods. One species of amphipod, *Grandidierella japonica*, appeared to be tolerant of elevated DDT concentrations. Average mortalities for *E. estuarius* were 42, 30 and 24 percent in samples collected from the Lauritzen Channel, Santa Fe Channel and Richmond Inner Harbor, respectively. Only sites in the Lauritzen Channel contained sufficient DDT to account for mortalities. Interstitial water threshold concentration of DDT was 0.5 µg/L, with 10-day LC50 of 2.2 µg/L. Pinza et al (in #23) found toxicity to *Rhepoxynius abronius* in sediment samples collected from the southeastern bank of the Richmond Inner Harbor Channel. Data from both studies suggest that contamination and associated toxicity were patchy; sites in Lauritzen Channel only meters apart produced 35 - 100 percent mortality. Similarly, samples collected from 26 sites in the South Bay produced an average 45 percent mortality in *R. abronius*, with a range of 20-100 percent mortality.

Long et al. (1990) #34 evaluated toxicity of sediments from San Francisco Bay with a variety of species. Three samples were from Oakland Inner Harbor (OIH), and three each from Yerba Buena, Vallejo and San Pablo. Tests included elutriate tests with mussel *M. edulis* and sea urchins *S. purpuratus*, solid phase sediment tests with amphipods *R. abronius* and *A. abdita*, and pore water tests with the polychaete *Dinophilus gyrociliatus*. Reduced survival (≤ 45%) was seen with *R. abronius* in all three OIH samples, 1 of 3 Vallejo samples and 1 of 3 from San

Pablo. The response in samples from Yerba Buena was more uniform and averaged about 65 % survival compared with 95% in controls. With *A. abdita*, only one of the OIH samples reduced survival and no effects on survival were seen in samples from the other sites. With *M. edulis*, reduced survival was seen in all of the samples from OIH, 2 of 3 from Yerba Buena, 1 of 3 from Vallejo and 1 of 3 from San Pablo. Larval abnormalities were also evaluated with this test; they were less than 25 % in all samples. With the sea urchins, there were no effects on normal development, but mitotic aberrations, micronucleated cells, and cytologic abnormalities were elevated in samples from all sites compared with the controls. With the polychaete, none of the samples reduced survival, but eggs per female were reduced in 2 of 3 of the samples from OIH and in all of the samples from Yerba Buena. Chemical analyses indicated that PAHs, DDT, total chlorinated pesticides, and PCBs were elevated in OIH samples compared with other sites. Vallejo sites had lower PAHs than other sites. All of the Bay sites had higher levels of these contaminants than samples from Tomales Bay. Correlations with different contaminants suggested that different organisms often responded to different contaminants.

Ankley et al. (1992) #42 point out the contribution of ammonia and hydrogen sulfide to sediment toxicity.

Species Effects

It is generally accepted that numerous fish, invertebrate, and algal species found in the Delta have declined in abundance over the past 20 years (). These declines have been attributed to a number of causes, including reduced outflows, increased diversions, and introduced species, which, in turn, have led to lower overall productivity in the Delta. Although adverse effects associated with toxic pollutants has been suggested as potentially contributing to the decline of species in the Estuary, comparatively little work has been done on species found in the Delta with respect to the effects of toxic substances. Studies that pertain to species of interest are described below.

Invertebrates

Neomysis mercedis: [add description of importance of opossum shrimp to estuary] Bailey

D-037225

(1993) evaluated the acute and chronic toxicity of the rice herbicides molinate and thiobencarb to *N. mercedis.* The data indicated that the two herbicides were additive in toxicity. Furthermore, comparison of measured concentrations of these pesticides in the Delta in 1985 with chronic toxicity values suggested that these pesticides may have reached toxic levels. Increased on-field holding times reduced concentrations of these two pesticides in subsequent years. However, in preceeding years, it is likely that concentrations were even higher than in 1985. Although no analytical data are available for the Delta for the years prior to 1985, the known application levels, shorter on-field holding times, lower river flows, and higher measured concentrations in Colusa Basin Drain, all would have contributed to higher concentrations in the Delta. As an example, in excess of 5 chronic TUs could have been reached in the Delta during the 1982 rice season (Bailey 1993).

Bailey et al. (1994) evaluated the sensitivity of *N. mercedis* to samples from Colusa Basin Drain. Ten of the 14 samples collected from CBD during rice season in 1989 produced complete mortality within 24 hr. Follow-up testing in 1990 indicated that the primary cause of toxicity was the organophosphorous pesticide methyl parathion (Finlayson et al. 1993). Reduced toxicity was observed in 1991 and was attributed to increased on-field holding times for this pesticide (Bailey et al. 1994b). In a hazard assessment on methyl parathion prepared by California Department of Fish and Game, Menconi and Harrington (1992) pointed out that cladocerans and mysids, two important components of the Delta food chain, were among the most sensitive species to this pesticide. They also pointed out that, based on measured concentrations in CBD and flows in the Sacramento River, concentrations of this pesticide could have exceeded the current water quality guidelines in the Delta in the early 1980s before stricter regulations were enforced. Their calculations show that nearly 1 µg/L methyl parathion could have been present in the Sacramento River downstream of Sacramento for up to four weeks. This value is five times higher than the 96-hr LC50 for *N. mercedis.* In 1988, analytical measurements made in the Sacramento River near Sacramento, showed that concentrations as high as 0.32 µg/L existed; this value exceeds the LC50 for *N. mercedis* by a factor of 1.5. Note that these concentrations would likely have reached the Delta largely unchanged since any further dilution would have occurred primarily through the relatively slow process of tidal mixing (Kuivila)

Testing of individual pesticides, including carbofuran, diazinon, and chlorpyrifos resulted in the following toxicity values for *N. mercedis* (Bailey et al. in prep., Brandt et al. 1992):

Pesticide	96-hr LC50 (µg/L)	NOEC (µg/L)
carbofuran	2.7-4.7	not reported
chlorpyrifos	0.07-0.09	0.04-0.05
diazinon	1.2-1.9	0.61-0.66

Monitoring data from different sites in the Delta suggests that these values are exceeded regularly, with the frequency depending on the pesticide. For example, Foe and Sheipline (1993) reported diazinon values in 26 samples collected from waters entering the Delta in January and February 1992. Concentrations in seven of the samples exceeded the LC50 for *N. mercedis* and 12 of the samples exceeded the acute NOEC.

Collectively, the data suggest that pesticide concentrations in the Delta regularly exceed levels associated with acute and chronic toxicity to this species, not to mention their respective water quality criteria. Although the identity of the pesticides has changed somewhat over time, the available data suggest that this situation has existed since the mid-1970s.

Bivalves: Pereira et al. (1992) #45 reported on bioaccumulation of hydrocarbons in the introduced clam *Potamocorbula amurensis* in Suisun Bay. Bioaccumulation of sediment hydrocarbons originating from petroleum sources was identified as was accumulation of polycyclic aromatic hydrocarbons derived from combustion. This species is a food source for a number of species including sturgeon and diving ducks.

Luoma et al. (1990) #52 evaluated temporal variation in trace metals in the introduced clam *Corbicula* in Suisun Bay and near the mouth of the San Joaquin River over a three year period. The authors concluded that there was little chronic contamination associated with Ag, Zn, or Pb, but that substantial chronic contamination was present in Suisun Bay with respect to Cu, Cd, and Cr. Inputs of Cr were dominated by discharges from a local steel mill, and Cu appeared to originate primarily from the Sacramento River during high inflows to the Bay. Sources of Cd were attributed to both riverine and local sources. The condition factor of clams in areas with highest contamination was reduced as was the abundance of larger clams. The data suggested that the bioavailability of Cu and Cd to the clams was greater in Suisun Bay than reported in other estuaries. In fact, tissue concentrations of Cu from Suisun Bay were 6-10 times greater than reported in un-enriched systems and tissue concentrations of Cd were found that exceeded levels reported anywhere in the literature. Some of the Cd tissue concentrations exceeded guidelines for human health consumption. Cr concentrations equal to the highest literature values found were also encountered.

Leland and Scudder (1990) #53 looked at tissue metals concentrations in *Corbicula* in the San Joaquin River. Selenium concentrations varied directly with soluble Se in riverwater. Se entered the system through subsurface drain and irrigation tailwaters. Elevated concentrations of Hg, As, Cu, Cd and Ni were also found, although the concentrations varied with respect to location and source. Boron and molybdenum were not accumulated and Cr, Pb, Ag, V, and Zn exhibited little geographic variability in tissue concentrations. The authors concluded that there was no evidence of synergism or antagonism between As, Cd, Cu, Hg, Ni, and Se with respect to their uptake. Based on their data, the authors found that available Cd, Cu, and Ni were not enriched compared with other sites. Hg was elevated in the tributaries and one site in the lower San Joaquin River, and Se was elevated primarily in the southern San Joaquin River and in tributaries that drained the western side of the Valley. Arsenic was enriched in the San Joaquin river and tributaries. Johns et al (1988) in #53 reported Se concentrations in the western Delta and Suisun Bay were elevated compared with sites in the southern Delta and lower Sacramento River. These were attributed to industrial discharges (nine of the largest point source dischargers in the Bay/Delta system release directly into Suisun Bay (Gunther et al in #52). >The major contribution of the San Joaquin River to Delta metal loads (aside from effects on productivity, nursery areas, etc., within the San Joaquin R.) occurs in spring when discharge is high and export pumps not exporting.<

Crayfish: Holck and Meek (1987) in #79 reported the LC50 of the crayfish *Procambarus clarkii* exposed to the pyrethroid insecticide resmethrin to be 0.82 µg/L.

Naqvi and Newton (1991) #251 examined the toxicity of endosulfan (Thiodan) to the Louisiana Red Crayfish (*Procambarus clarkii*). Endosulfan exhibited a half life of 2+years in sandy loam soil. The data suggested effects at concentations as low as 2 µg/L, but the small sample size and high variability preclude separating means.

Tadpole Shrimp: Walton et al. (1990) #79 evaluated the toxicity of four pesticides to tadpole shrimp (*Triops longicaudatus*). The following 24-hr LC50s (in µg/L) were obtained : 4.0 (chlorpyrifos); 73.8 (fenthion); 0.084 (cypermethrin); and 0.7 (resmethrin). In field studies, the lowest effective concentrations were (in g/ha): 11 (chlorpyrifos-EC 4); < 56 (fenthion-EC 4); 1-3 (cypermethrin-EC 2.5); and < 28 (resmethrin-18%).

Fairy Shrimp: Mizutani et al. (1991) evaluated the uptake of lead, cadmium and zinc by fairy shrimp *Branchinecta longiantenna*. These shrimp inhabit temporary rain pools. Organisms tolerated 25 mg/L Pb for two days, but 1 mg/L Cd or Zn was lethal. At 15 mg/L Pb or 0.1 mg/L Cd or Zn, exposed organisms died within 6-8 days. *B. longiantenna* accumulated all three metals which may be a source of concern since this species can be a significant source of food for migratory birds. Also, brine shrimp *Artemia salina,* which are widely distributed in salt ponds and consumed by birds, accumulate Cd to 100 mg/kg (wet wt.) Jennings and Rainbow in #37.

Hydra: Fu et al. (1994) #89 compared the toxicities of industrial wastewaters to *Hydra attenuata* and fathead minnows. Of the 20 samples tested for acute toxicity, *Hydra* were more sensitive than fathead minnows to 16 samples (by factors of 1.1-5.5), equally sensitive to 2 samples, and less sensitive (by factors of ≤ 2.2) to two samples. The authors point out that *Hydra* were more sensitive to antimony than rainbow trout, fathead minnows, annelids, amphipods, and caddisflies.

Fish

Striped Bass: [more to follow; include description of use and importance] Heath et al. (1993) examined the sublethal effects of rice pesticides methyl parathion, carbofuran and molinate on larval striped bass. The authors concluded that methyl parathion and molinate reduced swimming performance and the effect persisted for at least 10 days after the 4-day exposures. The authors also suggested that pesticide effects on food organisms could reduce food availability for larval striped bass and, therefore, adversely affect larval swimming performance.

Bergerhouse (1993) #127 evaluated the effect of elevated pH, ammonia and salt on the survival of larval hybrid striped bass. Up to six hr, survival was not affected by pH values of ≤ 9.4. The threshold for mortality appeared to be between 8.7 and 9.2. The lethal threshold for 0.26 mg/L total ammonia nitrogen was between pH 8.38 and 8.75 for five day old larvae. These corresponded to unionized ammonia concentrations of 0.02–0.04 mg/L, respectively. The addition of 0.7% naCl did not affect 2 and 4 day larvae at different pH levels but did significantly increase survival of 13 and 20 day larvae exposed to elevated pH. The salt treatment also reduced the joint effect of ammonia and high pH.

Hall (1991) reviewed the effects of water quality and contaminants on early life stages of striped bass. Findings of particular significance are reviewed here. 80 and 90 percent mortality were noted in larvae reared for 4 or 5 days in hardness of 34.6 and < 60 mg/L, respectively. Larval survival was maximized at salinities of 1– 10 ppth. Suspended solids of 500 – 1000 mg/L increased larval mortalities while ≥ 200 mg/L reduced feeding efficiency compared with 75 mg/L. Cd, dieldrin and TBT were acutely toxic to larvae at approximately 1 µg/L. Increased salinity appeared to reduce toxicity, particularly of mixtures.

Reardon and Harrell (1990) #132 evaluated the acute toxicity of copper sulfate to striped bass fingerlings at varying salinities. These authors also cite Hughes (1971) who reported 0.1 mg/L copper sulfate to be LC50 to larval striped bass. For fingerlings, LC50 concentrations of copper sulfate were 2.68 – 7.88 mg/L over a range of salinities of 5 to 15 ppth. Tolerance increased as salinity increased.

Pinkney et al. (1990) #137 investigated the effects of TBT on the survival, growth, morphometry and RNA-DNA ratio of larval striped bass. Adverse effects on growth were seen at the lowest concentration tested, 0.067 µg/L, and 0.8 µg/L reduced survival.

Hall et al. (1988) #140 investigated the effects of water quality in the Chesapeake Bay on striped bass prolarvae and yearlings using on-site and in-situ studies. Survival of prolarvae was reduced in the Choptank River within 96 hr. Associated parameters that might have contributed to toxicity were: 36-48 mg/L hardness, 150 μg/L monomeric aluminum, 3 μg/L Cd, and 40 μg/L Cu. Histological evaluation of yearlings exposed to Choptank Ricer water for nine days revealed alterations in gill and liver.

Cashman et al. (1992) compared chemical contaminants in moribund and healthy striped bass. They concluded that the moribund fish were greatly contaminated by chemicals compared with healthy fish obtained from the Delta and Pacific ocean. The moribund fish were associated with annual die-offs of striped bass, sometimes including 100s to 1000s of individuals, that most often are observed in the Carquinez Straits. Liver dysfunction was the most obvious aspect of the affected fish, but kidney and intestine were also involved. The die-offs usually occur after spawning. Sampling showed DDT levels in liver that ranged between 16 and 31 ng/g liver. Alicyclic hexanes were present at 0-10 ng/g liver and Arochlor 1260 was found at 440-760 ng/g liver. Livers of most of the moribund fish exhibited a mottled appearance. In addition to liver lesions, pathology of kidney, intestine, thyroid and interrenal tissue was observed. Dialkyl phthalates were detected in moribund fish livers at 10-20 times the concentrations found in the controls. Concentrations averaged 12 - 184 μg/g liver for individual phthalates; cumulative phthalate concentrations were closer to 375 μg/g liver. Liver concentrations of triazines and molinate were similar for control and moribund groups, but thiobencarb concentrations (1.2 μg/g liver) averaged about 2 times the control groups; thiobencarb is metabolized in striped bass liver to a potent hepatotoxin. Aliphatic hydrocarbons were found at 0.2-2.7 μg/g liver. These values were higher than in the controls; in addition, the chain length in the controls was limited to C16-22, whereas moribund fish contained chain lengths of C16-34. The authors proposed that the contaminant body burden may contribute to the die-off, perhaps as a multiple stressor interacting with osmotic stress.

Fabrizio et al. (1991) compared PCB concentrations in striped bass from different stocks on the Atlantic Coast. PCB concentrations were measured in edible filets (one side of skin on but w/o scales). PCB concentrations were variable, but fish from the Hudson river stocks had higher probability of exceeding 2 mg/kg limit for consumption than more southern stocks. This probability increased with age and size. Fish of ≥ 5 yrs old were most affected. At 610 mm minimum size (575 mm fork length) it was predicted that many harvestable fish would exceed tolerance limit set by Food and Drug Administration. (FDA reduced the tolerance limit from 5 to

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2 mg PCB/ kg fish in 1984). The data suggeseted that the probability of exceeding the tolerance limit was 12 times greater in 7+ year old fish compared with 2 yr old fish. Although highly variable, the relationship with size was also significant. Concentrations of PCBs in some fish exceeded 10 mg/kg and both sexes exhibited contamination. Uptake occurred through the water column or by ingestion of contaminated prey. Water-borne exposure was most significant for young fish and occurred relatively rapidly, with equilibrium reached within two weeks. Because mixed stocks over winter in the Hudson River, all are exposed to waterborne PCBs. Alternative exposure occurs through the food sources, primarily clupeids (herring) which are locally high in PCBs. The authors proposed that fishermen may need to target smaller fish to reduce potential for harvesting fish that exceed the tolerance limit. However, the overall harvest rate would have to be reduced to compensate for higher mortality rates on younger fish.

These data provide a basis for comparing concentrations of contaminants in the Estuary to those shown to be associated with adverse effects in other systems. They also suggest contributing factors to toxicity, particularly low salinity. Thus, early life history stages and adults during spawning are probably the most vulnerable. Multiple stressors are likely; striped bass are wide-ranging and sub-units of the population can vary dramatically in exposure and body burden of toxicants (). A retrospective analysis suggests that the striped bass decline may have been related to increased use and discharge of pesticides used on rice. Such effects would be particular apparent during low flow years when the dilution capacity of the River was minimal.

Cyprinids: [introduce carp, cyprinids in general] Saiki and Schmitt (1986) (in #44) reported total DDT residues in samples from carp collected from the San Joaquin River that exceeded the National Academy of Sciences recommended safe level for the protection of fish-eating wildlife. These same samples exceeded the EPA estimated safe tissue concentration by factors of 8.7-14.7. Toxaphene tissue residues as high as 3.1 mg/kg were also reported. This tissue level exceeded the threshold for adverse effects by a factor of 6 (Eisler and Jacknow 1985) (in # 44).

Alam and Maughan (1993)#66 evaluated the acute toxicity of diazinon and malathion to juvenile carp. The 96-hr LC50s for diazinon were 3.43 and 4.97 mg/L. For malathion, values

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of 10.21 and 10.38 mg/L were obtained. Alam and Maughan (1992) #242 reported LC50 values of 0.3-1.0, 0.16-0.77 and 0.44-1.33 mg/L for Cu, Hg and Pb, respectively, for juvenile carp. Smaller fish were more sensitive.

Kaur et al. (1993)#65 evaluated the effect of industrial effluents on the viability of carp eggs. Effluents from tannery, vegetable oil productiion, fertilizer production, and paper mill were evaluated. No-observable effect concentrations were 0.0001, 0.0001, 0.01 and 0.1 percent effluent, respectively.

Kaur and Dhawan (1993) #241 evaluated the sensitivity of carp eggs, 7-day larvae and 20-day fry to carbaryl, carbofuran, malathion and phosphamidon in hard water. Concentations of ≤ 50, 10, 100 and 100,000 µg/L did not affect hatchability. The greatest sensitivity occurred prior to gastrulation and closure of the blastopore. Larvae exhibited enlarged yoik and pericardial sac, stunted tails, and vertebral column flexure; these are non-specific responses produced by a variety of toxicants, including metals, hydrocarbons, detergents and organochlorines. The authors concluded that "safe" values for the pesticides were 12, 3, 43, and 198 µg/L, under the conditions of exposure. Interestingly, early life stages of this species exhibited relatively high sensitivity to carbofuran. The "safe" concentrations is exceeded in the Delta on a fairly regular basis. Although difficult, similar studies would be worthwhile performing on eggs and larvae of other species.

Reddy et al. (1992) #67 evaluated the effect of fenvalerate, a pyrethroid insecticide, on the survival and AChE system of carp. The 48 hr LC50 was 30 µg/L. At 10 µg/L AChE was depressed and ACh increased relative to the controls. Of the tissues examined, the effect was greatest in the brain. The fish also exhibited sublethal effects and behavioral changes. Reddy et al. (1991) #75 reported that 10 µg/L fenvalerate also inhibited Mg- and Na-KATPases in the gill, brain, liver and muscle tissues. For Mg-ATPase, the liver showed the greatest effect, followed by the muscle, brain and gill. Conversely, Na-KATPase was most affected in the gill, followed by the brain, muscle, and liver. Effects were noted with exposure periods as short as 6 hr (the shortest period tested) and increased in magnitude as the exposure durations were increased to 48 hr.

Neskovic et al. (1993) #74 examined the acute and subacute toxicity of atrazine, a triazine herbicide, to carp. The 96-hr acute LC50 was 18.8 mg/L. Significant changes in the activities of the enzymes alkaline phosphatase (serum, heart, liver, and kidney) and glutamic-pyruvic transaminase (liver and kidney) were found at 1.5 mg/L, the lowest concentration tested, after 14 days of exposure. Some small changes in gill histology were also noted at this concentration.

van der Weiden et al. (1992) #76 evaluated the effect of sediment contaminated with polychlorodibenzo-p-dioxins (PCDDs), polychlorodibenzofurans (PCDFs) and polychlorobiphenyls (PCBs) on 7-ethoxyresorufin O-deethylase (EROD) activity and cytochrome P4501A content in carp. All parameters were elevated following a 12-week exposure and the effect persisted for at least 3 mos.

Blevins (1991) #93 described the results of an in vitro *Salmonella* assay using liver microsomal enzyme preparations prepared from carp collected from polluted and unpolluted habitats. When incubated with 2-aminofluorene, the number of revertants increased in relation to the degree of pollution the carp were exposed to. The data suggest that this assay may be useful for screening for polluted environments, particularly those contaminated with mutagenic or carcinogenic chemicals.

Ruiz and Lorente (1991) #179 evaluated the seasonal accumulation of DDT and PCB in carp muscle from the Ebro Delta in Spain. Body burdens were found to increase in spring when channels opened for rice culture, which resuspended sediment particles containing bound contaminants. Later, another peak was found that corresponded to current applications of pesticides. Decreases in body burdens were found following reproduction and shedding of lipid-loaded gametes.

Kozaric et al. (1992) #215 evaluated the effects of Cd on hydrolase activities in liver and kidney of carp using histochemical methods. Non-specific esterase activity was inhibited in liver cells and kidney tubular cells. Acid and alkaline phosphatase activities were elevated in liver, but not in kidney. Exposure was 30 days at 240 µg/L; whether these alterations would be apparent at lower concentrations was not established.

Tsuda et al. (1992) #217 evaluated uptake and excretion of different pesticides in carp. There was rapid accumulation and depuration. The BCF for chlorpyrifos was 460 and equilibrium was reached in three days. The excretion rate was rapid, with a half-life of 34.7 hr. Captan exhibited a longer half-life (69.3 hr), but a BCF of only 100, which was reached within 24 hr.

Sunderam et al. (1992) #247 tested various fish species against endosulfan. Carp were the most sensitive, with 96 hr LC50 of 0.1 µg/L. For rainbow trout, the LC50 was 0.7 µg/L (0.3 µg/L was reported elsewhere for trout). The LC50 for mosquito fish was 3.1 µg/L.

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Dyer et al. (1993) #88 investigated the synthesis and accumulation of stress proteins in fathead minnows *Pimephales promelas* exposed to inorganic arsenic (sodium arsenite). The 96-hr LC50 was found to be 9.9 mg As/L. The stress protein response was elicited rapidly, within 2 hrs in the gill at 25 mg/L As. In the gill, synthesis of 20, 70, 72, and 74 kD proteins were significantly correlated with mortality but, in striated muscle, only the 70 kD proteins were correlated with mortality. Gill tissue produced a greater variety of proteins and a greater response at lower concentrations than did muscle tissue.

Lindstrom-Seppa et al. (1994) #41 described the uptake of 3,3',4,4'-tetrachlorobiphenyl (TCB) and induction of cytochrome P4501A in fathead minnows. High concentrations suppressed EROD activity. TCB was passed into eggs. Maintenance of CYP1A may last 2 weeks following induction. In female fish, estradiol may suppress CYP1A induction. Endothelial cell lining was an important source of CYP1A, with greater induction than in epithelial structures.

Hermanutz (1992) #103 described malformations in fathead minnows reared in model ecosystems that contained waterborne Se at concentrations of 10 and 30 µg/L. Thus, Se was not only available in the water but also through dietary sources produced within the system. Fish were evaluated after approximately 1 year in the systems. Standing stocks were lower in the treated ecosystems than in the control systems and offspring exhibited a high incidence of externally visible malformations.

Changes in fathead minnow behavior were evaluated following 24-hr exposures to pesticides by

Peyster and Long (1993) #104. 24-hr LC50s for diquat and fenitrothion were 76.5 and 7.5 mg/L, respectively. Behavior was a more sensitive endpoint; changes were found at concentrations of 9.2 and 2.6 mg/L, for diquat and fenitrothion, respectively. These concentrations were the lowest tested; behavioral effects may also have occurred at lower concentrations.

Hall and Oris (1991) #115 evaluated the effect of anthracene on the reproduction of fathead minnows. Anthracene was accumulated by the fish and passed into the eggs. Larval deformities was observed as was reduced larval survivorship and decreased reproductive output. The data indicate that anthracene is toxic in absence of UV radiation. (data of PAHs in general given). Females had higher BCFs than males and reproductive output was reduced by up to 52 percent. The NOEC was < 6 µg/L in this study and ≤ 8.2 µg/L for daphnid reproduction (in absence of UV) as reported by Holst and Giesy (1989). The authors considererd that reproductive effects could be related to interference with binding of hormones to receptor sites or interactions between MFO induction and hormone levels. Based on concentrations in eggs (wet weight) the NOEC fell between 3.75 and 8 µg/g. (Good discussion of interactions between MFOs and potention repro effects.)

Weber et al. (1991) #116 evaluated the effects of a 4-week exposure to waterborne lead (0.5 and 1.0 mg/L) on the feeding abilities and neurotransmitter levels in juvenile fathead minnows. Lead exposed fish exhibited reduced ability to feed on live prey. Brain levels of serotonin and norepinephrine were increased following exposure, but there was no change in dopamine concentration.

Lien and McKim (1993) #117 evaluated the uptake of 2,2',5,5'-tetrachlorobiphenyl in fathead minnows. The authors found that uptake across the skin was similar to that across the gills and attributed this to the relatively large surface to volume ratio compared to larger fish and the comparatively small diffusion distance, again compared to larger fish.

Welsh et al. (1993) #121 investigated the effects of dissolved organic carbon and pH on the toxicity of Cu to larval fathead minnows. Toxicity was inversely related to DOC and pH. The

LC50s ranged from 2 (pH 5.6, DOC 0.2 mg/L) to 182 µg/L (pH 6.9, DOC 15.6 mg/L). A predictive equation that incorporated DOC and pH explained 93 percent of the variation in toxicity within a pH range of 5.4 - 7.3 and DOCs of 0.2 – 16 mg/L. The toxicity of Zn was also reduced by the formation of organic complexes.

Hickie et al. (1993) # 122 evaluated the toxicity of trace metals to larval rainbow trout and fathead minnows. Below pH 4.9, Al was the dominant toxic cation while Cu was predominant at pH 5.8. Rainbow trout were less sensitive to low pH and metal ions than the fathead minnows.

Hartwell et al. (1989) compared toxicity and avoidance response to five metals with golden shiner. 96-hr LC50s for Cr, Cu and As were 55, 84.6 and 12.5 mg/L; shiners exhibited avoidance response at 2 orders of magnitude lower concentrations. No avoidance response was observed with Cd (up to 68 µg/L) and Se (up to 3.5 mg/L). LC50s for these metals were 2.8 and 11.2 mg/L, respectively. Water hardness was 72 mg/L.

Chiasson (1993) #124 measured avoidance responses in golden shiners to suspended sediment concentrations of 15, 75 and 150 JTU. Responses were noted at 75 and 150 JTU.

<u>Salmonids</u>: [describe use as migratory pathway, juvenile feeding] Servizi and Gordon (1990) #134 evaluated the toxic interactions between ammonia and suspended sediment with juvenile chinook salmon. The LC50 of suspended sediment was 31 mg/L and the LC50 for ammonia was 0.45 mg/L, as un-ionized ammonia. Joint toxicity was slightly less than additive.

Short and Thrower (1987) #131 investigated the toxicity of TBT to juvenile chinook salmon. The 96-hr LC50 was 1.5 µg/L. TBT concentrations in the brain, liver and muscle were 200 – 4300 times higher than the concentration in water. The linear relationship between exposure time and mortality suggested that longer exposures would result in greater mortalities at lower concentrations.

Pinkney et al. (1990) #137 found that 0.2 µg/L TBT reduced growth in rainbow trout fry and 0.09 µg/L reduced growth in silversides. Both values were lowest concentrations tested.

Mitchell et al. (1987) #133 investigated the acute toxicity of the herbicides Rodeo and Roundup to rainbow trout, chinook, and coho salmon. For Roundup, based on total formulation, the 96 hr LC50s ranged between 15 and 22 mg/L. Based on glyphosate, the active ingredient, the LC50s ranged from 7.4 to 12 mg/L. Rodeo was less toxic; based on total formulation, the LC50s ranged between 680 and 1440 mg/L. Based on glyphosphate, the LC50s were 130 to 290 mg/L. Coho salmon smolts challenged with seawater were not affected by up to 2.78 mg/L Roundup, based on total formulation (Chapman #135).

Hamilton and Buhl (1990) #136 investigated the effects of arsenate, arsenite, cadmium, copper, mercury, silver, vanadium and zinc on chinook salmon fry. These metals are associated with discharge from the San Luis Drain that collects irrigation waters from the westside of the San Joaquin drainage. In terms of toxicity, Cd and Cu > Hg > Zn > Vd > arsenite > arsenate > chromium (no definitive tests were completed with silver due to precipitation). By comparing toxicity to actual concentrations found in the Drain, Cd, Cu, Hg, and Zn were estimated to be of concern in fresh or brackish receiving waters. LC50s for the first three metals ranged from 17 – 101 µg/L, whereas the LC50 was 1.3 – 2.9 mg/L for Zn. Toxicity continued to increase over the 96-hr exposure period, particularly for Cd, suggesting that longer exposure would be associated with increased effects. In most cases, brackish water appeared to reduce toxicity. Mixtures of the metals exhibited approxiamtely additive toxicity.

Hamilton et al. (1990) #138 evaluated the effect of dietary selenium on chinook salmon. Survival was reduced in fish fed ≥ 9.6 µg/g Se and growth was reduced in fish receiving ≥ 5.3 µg/g Se. The selenium was introduced by incorporating mosquito fish, obtained from San Luis Drain, that contained high Se levels into the fish meal component of the diet. Selenium was not concentrated in the salmon to levels greater than in the diet.

Ictalurids: Lin et al. 1994 #81 looked for metabolites of polycyclic aromatic hydrocarbons (PAHs) in the bile of brown bullhead *Ameiurus nebulosus* collected from four tributaries to Lake Erie. Concentrations of PAH metabolites in the bile of fish from sites with contaminated sediments were 5-20 times greater than those from sites with uncontaminated sediments.

Murdoch and Hebert (1994) measured mitochondrial DNA diversity in brown bullhead from sites in the Great Lakes having sediments contaminated with heavy metals, PAHs, PCBs, chlorinated pesticides and petroleum hydrocarbons. Their results demonstrated considerable differences in mDNA between sites, with a consistent reduction in haplotype diversity at contaminated sites compared with fish sampled at reference sites. The authors concuded that the decreased diversity was associated with population bottlenecks, in this case severe selection associated with environmental degradation, i.e., pollution. They also pointed out the advantages of brown bullheads for this type of research; they are sensitive to contaminants frequently found in dediments and tend to be representative of local conditions because they do not undergo extensive migrations.

Steward et al. (1990) #86 investigated the metabolic fate of benzo[a]pyrene in brown bullhead. Most of the BP was found in the bile, liver, and gut, with significant quantities also associated with the spleen, gonads and muscle. The hepato-biliary system was the major route of excretion. Metabolism produced the highly genotoxic BP-7,8-diol and other bioactive metabolites. The presence of the parent compound and active intermediates in the muscle is of concern to those who consume contaminated fish.

Hasspieler et al (1994) #31 compared glutathione response against xenobiotics in channel catfish and brown bullhead. Brown bullhead mounted less of a response and maintained lower levels of hepatic total glutathione and reduced glutathione than channel catfish. This is consistent with brown bullhead expression of neoplasms in contaminated systems compared with channel catfish which rarely express pollutant mediated neoplasia. Thus, brown bullhead would appear to be more sensitive to GSH arylators and oxidants. Reduced glutathione protects against oxidants as an antioxidant defense and is a substrate for conjugation reactions.

Gallagher and Di Giulio (1989) #38 evaluated the effects of complex waste mixtures on hepatic monooxygenase activity in brown bullhead. In spite of lip and jaw lesions and liver damage, measures of MFO activity (cytochrome p450, EROD, etc.) were not good indicators of fish from the contaminated site. The authors pointed out that MFOs may respond well to specific chemicals (PAHs and PCBs) but their response to complex mixtures is not well characterized; the presence of selected metals may suppress enzyme activity.

and kidney of brown bullhead exposed to the PAH 3-methylcholanthrene. After 7 days, treated fish showed increased activity of aryl hydrocarbon hydroxylase in liver and kidney, with little effect on epoxide hydrase or glutathione-S-transferase. This may be of concern because the secondary enzymes that detoxify the oxidative products were not induced in the same time course as the oxidizing enzymes, potentially leaving increased levels of arene oxides or epoxides available to interact with cellular components. Microsomal preparations from liver and kidney treated with benz(a)pyrene showed that liver microsomes converted a greater proportion to BP -quinones than kidney preparations. The quinones are known to cause DNA damage and may be related to the susceptibility of brown bullhead to liver tumors. Interspecific differences in formation of quinones was noted and may be a reason for differences in species susceptibilities.

Gallagher et al. (1992) #110 investigated the acute toxicity of a fungicide chlorothalonil to channel catfish. Toxicity was increased 3X (120 vs 40 µg/L) following depletion of liver and gill glutathione (GSH). Exposure to chlorothalonil induced GSH levels in liver and gill. Data indicated that exposure to multiple toxicants may increase toxicity as one or more of the chemicals deplete tissue GSH levels.

Sikka et al. (1990) #112 compared the metabolism of benzo[a]pyrene, a polycyclic aromatic hydrocarbon, in hepatic microsomes from brown bullhead and carp. BP was metabolized over 10 times faster in carp microsomes than in brown bullhead microsomes. Similar metabolic products were obtained, but carp produced a much greater proportion of benzo-ring dihydrodiols compared with brown bullhead, which produced largely BP-phenols and BP-quinones. These results may explain the greater propensity for brown bullhead to form epidermal and hepatic tumors.

Plumb and Areechon (1990) #233 examined the effect of a 30-day exposure to malathion on the immune response of channel catfish challenged with the pathogen E. ictaluri. 1.74 mg/L malathion reduced the antibody response by 85 percent compared with the control; 0.5 mg/L reduced the response by 20 percent compared with the control. Deformities (80 and 10 percent, respectively) and behavioral changes were also noted at both concentrations.

<u>Mosquitofish</u>: Lee et al. (1992) #72 evaluated the effect of acute inorganic mercury exposure

on populations of mosquitofish. There was a maternal effect in that groups of fish that shared a common mother exhibited similar sensitivities to Hg. This implies a heritable genetic component to sensitivity.

Chagnon and Guttman (1989) #90 evaluated the effect of copper and cadmium on the survival of populations of mosquitofish containing different allozyme genotypes. For both metals, differences in survival were associated with different genotypes. The authors noted that correlations between heavy metal stress and allelic and genotypic frequencies at the phosphoglucomutase (PGM) and/or the glucose phosphate isomerase (GPI) loci have been generally noted in marine and freshwater invertebrates and fish. Strittholt et al. (in #90) suggested that the current lack of allozyme variability in yellow perch in Lake Erie was due to selection from heavy metal pollution.

Heagler et al. (1993) #91 also investigated the effect of mercury exposure on allozyme genotypes in mosquitofish. One of the nine loci investigated, GPI, was correlated with time to death in the laboratory study. A follow-up investigation compared fish from Hg-contaminated and control sites. The fish from the contaminated site exhibited significantly lower frequency of one of the GPI alleles than did fish from the uncontaminated site.

In follow-up work, Kramer and Newman (1994) #92 compared the effect of Hg on the gluconeogenic properties of preparations of two different GPI allozymes. The results suggested that the allozyme generally associated with increased sensitivity to Hg (and As), $Gpi\text{-}2^{38/38}$, was not differentially inhibited by Hg, suggesting that the associated allele was a marker closely related to a gene(s) that confers susceptibility to Hg toxicity.

Tietze et al. (1991) #126 evaluated the toxicity of chemicals used in control of mosquitos to mosquito fish. The most toxic material was resmethrin, with a 24-hr LC50 of 7 µg/L.

<u>Centrarchids</u>: McCloskey and Oris (1991) #107 investigated the toxicity of anthracene, a model polycyclic aromatic hydrocarbon, to bluegill. Potential interactions with toxicity and temperature and dissolved oxygen levels were observed. Toxicity was increased in the presence of UV light; toxicity was not expressed up to 35 µg/L in the absence of UV light. In the presence

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of UV light, 96-hr LC50 concentrations ranged from 1.3 – 8.3 µg/L.

Atherinids: Middaugh et al. (1991) #231 examined the effect of water contaminated with PAHs on toxicity and teratogenicity in silversides. Exposure to contaminated sediments near discharge site resulted in rapid increase of liver cytochrome P-450-IA1 in juvenile fish. Liver and skin lesions were present in spot collected from the discharge site. Effects were observed in *Menidia* embryos exposed for ten days until hatch at concentrations of 0.28 – 2.1 mg/L, while no effects were observed at concentrations of 0.028 – 0.15 mg/L. Ultrafiltration reduced the concentration of PAHs by almost a factor of 100. In general, PAHs' rate of decomposition is almost constant, while anthropogenic inputs, such as oil products and spills, burning, and oil refining, have increased. Eagle Harbor contain up to 6.5 g/kg dry wt PAHs in sediment and exposure produced liver lesions in fish. Liver lesions are also present in fish (*Fundulus*) in the Elizabeth River which averages 2.2 g PAH/kg dry sediment.

Hemmer et al (1992) #234 compared the sensitivity of larval topsmelt and silversides to a variety of chemicals using static 96-hr tests and 7 - 33 day old fish. LC50s were within a factor of 2 for most chemicals but varied up to a factor of 6.7. Seven-day old fish were approximately 20 percent more sensitive than 28 day old fish. In comparisons with other species, bluegill were generally more sensitive (average 3X), trout next (average 2X), and fathead minnows quite a bit less sensitive (from 23X up to 63X). Reproducibility within laboratories for these tests was 1.4-2; reproducibility between labs was within a factor of 4.4. The authors also noted that fathead minnows were particularly insensitive to OPs.

Other Fish: Noga et al. (1991) #216 described dermatological disease in fish from the Tar-Pamlico estuary in North Carolina. Ulcerative mycosis affected a variety of species including striped bass, flounder. Only one tumor was found. While the causes were not determined, the mycosis appeared to primarily affect fish found in intermediate salinities, rather than high salinities or in freshwater. In general, skin ulcers are indicative of polluted environments. (Sindermann 1990).

Vittozzi and De Angelis (1991) #226 reviewed acute toxicity data on fish. Fish species included rainbow trout, fathead minnow, bluegill, carp, medaka (check Oryzias latipes) and zebrafish. 200 chemicals were evaluated; approximately 10 % were phosphoesters, including diazinon, chlorpyrifos and malathion. Carbamates were also represented. No relationship between n-octanol water partition coefficient and toxicity was found. Evidence for species dependent toxicity was found; differences between species varied by factors of 10 to 300. Organophosphorous chemicals (OPs) generally accounted for most of the species selective toxicity observed in the data set. Fathead minnows were the most resistant species in virtually all comparisons. OPs were generally 10 – 300 times more toxic to bluegill than to fathead minnows. OPs also exhibited greater toxicity to trout than to fathead minnows; up to 160 times more sensitive. Differences in sensitivity may be due to differences in sensitivity of cholinergic system and/or development of detoxification systems. In general, since approximately 4 of 100 chemicals shows species dependent toxicity, use of one species to assess effects is not adequate. However, this frequency is much higher for organophosphorous chemicals. Therefore, one could consider the use of safety factors; a safety factor of 100 applied to fathead minnow data would include all but the three most selective OP chemicals.

Thybaud (1990) #224 investigated differences in toxicity between lindane, a chlorinated hydrocarbon, and deltamethrin, a new generation pyrethroid. Lindane exhibited high bioaccumulation but lower toxicity. Deltamethrin was rapidly metabolized. Deltamethrin approximately 100 to 1000 times more toxic to *Rana* tadpoles and *Gambusia* than lindane.

Clearance and uptake rates and total amount of contaminant accumulated may change with change in salinity. With higher salinities, clearance rates increase and uptake rates decrease with pentachlorophenol in killifish (*Oryzias latipes*). Freshwater-acclimated fish accumulated more PCP than saltwater fish (Tachikawa and Sawamura 1994) #109.

Birds

<u>*Double-Crested Cormorant (Phalacrocorax auritus)*</u>: Jones et al. (1994) #78 evaluated the bioamagnification of PCBs and TCDD-equivalents in eggs and chicks of the double-crested cormorant at different sites in the Great Lakes. Toxic concentrations of these materials results in induction of mixed function oxidases, depletion of hepatic retinoids and vitamin A, porphyria, edema and wasting syndrome. Similar effects were also seen in salmonid fishes and mink in the

Great Lakes. The biomagnification factor from forage fish to cormorant eggs was 31.3. Concentrations in the chicks decreased immediately following hatching, then increased in proportion to the mass of fish consumed. Once the chicks initiated thermoregulation, the rate of accumulation increased. Weathering actually increased the proportion of more toxic PCB congeners compared with the original Arachlor mixture. Concentrations in chicks were more closely associated with local conditions than concentrations in eggs; adults tended to mix and forage widely when not feeding young. Foraging for young usually occurred relatively close to the nesting site.

Cormorants are present in the Estuary where they occupy a similar niche high in the food web. Therefore, they would be a good indicator species to evaluate trophic effects of contaminants. By determining contaminated components of the forage base, insights should be gained regarding sediment and/or water column contaminants of concern.

Bald Eagle: Bald eagles occupy high trophic levels and so may be contaminated by xenobiotics that accumulate in the food chain to harmfuls levels. They may also be adversely affected by ingesting lead fragments in dead or wounded waterfowl (Langelier et al 1991 #61 and Gill and Langelier 1994#60) or by accidental pesticide poisoning (Bowes et al 1992 #62, Colburn 1991 #63). Pesticides associated with eagle mortalities include dieldrin, endrin, DDE, DDT and carbofuran. Bald eagles are present in this system seasonally and could be affected by lead-contaminated waterfowl during hunting season, although there is little local information on this. Data from bald eagle populations in the Great Lakes suggests that accumulation of toxic chemicals through the food chain has reduced successful reproduction in several populations. The chemicals of major concern include DDT, DDE and PCBs, dioxins and furans (Colburn 1991 #63). The problem is exacerbated because the eagles not only feed on contaminated fish, but also on birds that feed on the fish, thus effectively increasing the biomagnification. Other birds feeding on the same types of feed also are experiencing reproductive failures, including Caspian, Foster's, and common terns, osprey, double-crested cormorant, and herring gull. Since the eagles are transient to the Bay/Delta, looking at some of these fish-eating birds that are residents would be appropriate. Mink and otter were also indicated as species showing reproductive failure and population declines consistent with the eagle in the Great Lakes system.

Grubb et al (1990) #20 reviewed the relationship between eggshell thinning and contaminant levels in bald eagles in Arizona. Eggshells were still thinner than from pre-DDT era, but productivity was improving. All eggs analyzed included detectable levels of Hg, DDE and PCBs. Hg was below levels known to cause effects. Concentrations of contaminants (DDE and PCBs) decreased in eggs between 1977 and 1982. Overall, DDE levels of 1-2 ppm wet weight in eggs did not appear to affect reproductive success and neither did PCB levels of 0.4-0.9 ppm. Local populations varied appreciably in their tissue concentrations.

Anthony et al (1993) #24 described environmental contaminants in bald eagles in the Columbia river estuary. High levels of DDE, PCBs and TCDD were found in eggs and adults. DDE and PCBs were also found in nestlings indicating early dietary exposure. Hg levels were higher in adults, indicating accumulation with age. The role of dioxin was unclear but concentrations in eggs were similar to those found to have deleterious effects in other species in laboratory exposures. Resuspension of dredged river sediments played an important role in bioavailability. Eggshell thinning was present and related to DDE and PCBs. DDE and PCB concentrations averaged 9.7 and 12.7 ppm (wet weight) respectively. TCDD concentrations averaged 60 ppt in eggs. Contaminant levels were 2-3 times higher in northern squawfish than in suckers; both fish are components of eagles' diets. Fish-eating birds were probably the source of DDE, but fish were more likely source of PCBs (tissue concentrations of PCBs in all fish samples exceeded 0.5 ppm - the recommended level for protection of fish-eating birds and mammals). Three of 12 fish samples contained Hg concentrations that exceeded dietary levels shown to interfere with successful reproduction in mallards. Kubiak et al in #24 concluded that PCBs were responsible for embryotoxicity in Foster's tern in Lake Michigan. The TCDD residues in eagle eggs (60 ppt) were higher than concentrations (37 ppt) found to impair reproduction in Foster's terns in Lake Michigan. TCDD levels in prey species averaged 2.8 ppt, 40 times higher than the fish consumption guideline for human health (EPA). Elevated concentrations of DDE and PCBs were also found in mink, otter, and black-crowned night herons. A link between fish uptake and dredging of PCBs and DDE has been demonstrated (Seelye in #24).

Frenzel and Anthony (1989) #25 found that exposure to environmental contaminants in wintering bald eagles was greatly dependent on the exposure history of the prey item. In many cases, exposure to embedded lead shot in waterfowl constituted the greatest hazard. Organochlorines and Hg were low in voles and jackrabbits. Waterfowl had higher levels, with dabbling ducks < diving ducks < western grebes and California gulls. Dietary exposures of wintering and nesting and rearing eagles may differ greatly.

Wiemeyer et al (1993) #30 reviewed the effect of environmental contaminants on bald eagle productivity. Young production was normal if the eggs contained < 3.6 μg DDE/g, was reduced by half between 3.6 and 6.3 μg/g, and reduced by half again at concentrations > 6.3 μg/g. Other contaminants were also associated with poor reproduction but, since they were highly correlated with DDE, it was difficult to assess their individual effects. Concentrations of contaminants appeared to be declining in many parts of country between mid-70s and mid-80s, but specific contaminant(s) varied with location. The data support the concept of a threshold effect for DDE. The effect of PCBs is uncertain, there are very toxic coplanar congeners that we know relatively little about.

Craig et al. (1990) #36 evaluated lead toxicity in golden and bald eagles. Liver concentrations of ≤ 2 ppm were indicative of uncontaminated birds, 2-8 ppm, sublethally contaminated, and ≥ 8 ppm acutely contaminated. Uptake of lead was probably not from organisms contaminated from feeding at a lead-contaminated site, but from hunter-killed game, including waterfowl.

Kozie and Anderson (1991) #95 reported that bald eagles nesting on the shores of Lake Superior exhibited lower production than eagles from inland sites in Wisconsin. Contaminant levels were also higher in eagles from Lake Superior sites compared with inland sites. Herring gulls were considered the primary source of contaminants and contained much higher DDE and PCB residues than fish (5.5 vs 0.07 μg DDE/g and 16.95 vs 0.21 μg PCB/g wet weight, respectively). DDE contamination in Lake Superior appeared due primarily to atmospheric sources. Concentrations in fish appear to be declining; in the late 1960s concentrations averaged 0.46 μg/g. Concentrations ≥ 2.8 μg DDE/g in diet contributed to reproductive failure in bald eagles.

Bald eagles are mostly transient in the Estuary. However, they are affected by many of the same contaminants that affect other birds high in the food web. Thus, monitoring other key indicator species, such as terns, gulls, grebes, cormorants, and so on, should provide a valuable basis upon which to compare trophic effects with those identified in the Great Lakes and elsewhere.

California Black Rail: Evens et al (1991) #26 reviewed the status of the California black rail. The bulk of the population was confined to the northern marshlands of San Francisco Bay. The populations are undergoing a decline, presumably due to habitat loss or degradation. The effects of contaminants are not known, but have affected other bird species in estuary.

Willet: Custer and Mitchell(1991) #245 evaluated contaminants in willets (*Catotrophorus semipalmatus*) collected at outlets of two agricultural drains in Texas. Elevated arsenic levels in liver were noted but concentrations (max 15 ppm dry wt.) were < than associated with acute toxicity. Se was at background concentrations, Hg concentrations were elevated (2-17 ppm), but were generally less than associated with mortality. However, Hg concentrations could be associated with reproductive effects. Cholinesterase in brain did not indicate recent exposure to OPs. However, ChE depression was observed in the past and could have been associated with recent application of OP or carbamate insecticides. Only arsenic was elevated in the agricultural drains. The authors indicated that body burdens of 12-68 ppm DDE are dangerous to birds.

Blackbirds: Meyers et al. (1992) #80 evaluated the toxicity of chlorpyrifos and dimethoate to red-wing blackbirds *Agelaius phoeniceus* and starlings *Sturnus vulgaris*. For dimethoate, the reported LC50 values for adult starlings and blackbirds were 32 and 6.6 mg/kg, respectively. For chlorpyrifos, the values were 5.0 and 13.0 mg/kg, respectively. Nestlings responded differently. A single dose (oral) of 2 mg/Kg chlorpyrifos reduced the survival of blackbird nestlings by approximately 50 percent over a 10 day period, but did not affect survival of starling nestlings. In contrast, a single dose of 50 mg/kg dimethoate did not affect the survival of blackbird nestlings, but reduced the survival of starling nestlings by 56 percent. Growth was not affected in birds that survived exposure.

Mammals

<u>Mink and Otter</u>: Mink have been widely used in toxicology as a model species for drug and metabolism studies (Calabrese et al. 1992 #56, Bursian et al. 1992 #55). They are also widely distributed throughout the Delta where they are one of the top mammalian predators. Consequently, they should reflect contamination frequently associated with a position near the top of the food chain. Calabrese et al (1990) reported that mink are highly sensitive to PCBs. These authors noted that mink were not particularly sensitive to chlorinated hydrocarbon insecticides but that adverse effects on growth and reproduction were found when mink were fed PCB-contaminated fish from the Great Lakes. Methyl mercury in fish also caused neurological toxicity and dioxin induced wasting syndrome and gastric lesions. Aulerich et al 199? #58 reported that diet containing > 12.5 mg/kg heptachlor given for 28 days resulted in adverse effects, particularly on growth. This level of sensitivity was considerably greater than for rodents given dietary heptachlor. Consideration should be given to a monitoring program that looks at body burdens in mink to evaluate the potential for adverse effects due to accumulation of xenobiotics.

Ropek and Neely (1993) #21 reported Hg concentrations elevated (3-4X) in otters compared to the concentrations in their diet. Hg was accumulated in both liver and kidney (higher in liver). Males accumulated more than did females. Average levels were 2.2 mg/Kg in liver (dry wt.) and 1.5 mg/Kg in kidney. Daily dietary levels of 2 ppm methylmercury were lethal within 7 months (O'Connor and Nielsen 1981 in #21). Liver levels in otter from polluted areas (Georgia) averaged 7.5 ppm.

<u>Muskrat</u>: Halbrook et al. (1993) #94 compared muskrats inhabiting polluted and unpolluted sites. They found that muskrats inhabiting a contaminated waterway exhibited reduced fat indexes and spleen weights, greater adrenal weights, and increased incidence of disease and parasitism compared with those found in relatively uncontaminated sites. Increased body burdens of aluminum, cadmium, copper, nickel, zinc, and polyaromatic hydrocarbon compunds were associated with muskrats collected from the polluted site. Fish from the same waterway also exhibited signs of adverse effects including fin erosion, cataracts, and liver tumors.

Interactive Effects

Barnthouse et al. (1990) evaluated the effect of life history, data uncertainty and exploitation intensity on the potential effects of toxic contaminants on fish. Their evaluation compared the responses of fish with two different life history strategies, gulf menhaden and striped bass. Ecological theory suggests that species with long life and low reproductive potential, such as striped bass, may be most vulnerable to changes in environment. However, risk assessment also depends on other data including results from different types of toxicity tests, tests on different species (taxanomic distance), and Quantitative Structure Activity Relationships. Fishing mortality overlays these effects as it reduces ability of populations to sustain themselves. Differences in life history strategy are apparent between the two species: menhaden are short-lived (≤5 yr) filter feeders, while striped bass are large, long-lived (≥ 10 yr) piscivores. Data from power plants suggests a factor of 2 precision in predicting efffects on year class abundance of fish population is possible, but this is probalby near the best we can do. Order of magnitude uncertainty in predicting contaminant mortality is probably more typical(see 17 in refs). Both test type and taxanomic distance are potentially responsible for significant uncertainty in predicting effects. For example, chronic tests on the species and contaminant of interest provide the greatest accuracy towards predicting effects; fecundity data are important since many toxicants exhibit reproductive effects and fecunditiy may be the most sensitive endpoint. Conversely, a life cycle test on another species or an LC50 on the species of interest increases the uncertainty to factor of 150. LC50 estimates on another species or QSAR are virtually useless for risk assessment since the uncertainty factor approaches 300 and may reach almost 3 orders of magnitude for predicting an EC10. The effect of fishing is to further reduce EC10 estimates, but only by factors of 6 -10, which are modest compared with the uncertainty of estimating contaminant risk (up to a factor of 300). Modelling indicated that striped bass would be more at risk than menhaden. Important factors usually ignored include effects on fecundity, cumulative effects of toxicants on different life stages, effect of life history on capacity of population to sustain additional mortality, and the combined effects of multiple stressors.

Marcogliese et al. (1992) #180 reported that the composition of the zooplankton was altered following modification of a fish community by selenium toxicity. In this case, the pisciverous species in a cooling reservoir were eliminated, leaving only planktivores. Although species numbers remained similar, abundance of the formerly predominant cladocerans and copepods declined, particularly those > 1 mm in body size. This appeared due to selective feeding by planktivorous fish and loss of refugia for zooplankton in areas of the reservoir formerly dominated by piscivores.

Brazner and Kline (1990) evaluated the effects of chlorpyrifos on the dietary composition and growth of fathead minnows reared in littoral enclosures in a natural pond. Chlorpyrifos was applied once at 0.5, 5.0 and 20.0 µg/L. The initial concentrations decreased by approximately 60-80 percent within the first 24 hr after application, but the rate of decay decreased thereafter. At the 0.5 µg/L concentration, the concentration was 0.11 µg/L after 96 hr and < 0.01 µg/L after 384 hr. Effects were apparent in all concentrations. In the lowest concentration, which has the greatest relevance to concentrations found in the Sacramento-San Joaquin watershed, reduced growth was observed 15 days after application (approximately 33 % less than controls). Reduced growth was attributed to reduced numbers of rotifers, cladocerans, immature hydracarina, chironimids and copepod *nauplii* compared with untreated ponds. Cladocerans and chironomids decreased by 1 – 2 orders of magnitude in abundance within four days of application. Copepods declined by approximately 50 percent and the response was less dramatic for rotifers. The authors pointed out the possible "costs" of reduced food and growth, including greater vulnerability to predation.

deNoyelles et al. (1989) #250 described the effect of atrazine on pond microcosms after 136 and 805 days of exposure. Phytoplankton production and the supported food chain suffered transient decrease to 50 percent of the control values, which lasted 3 weeks in duration. A change in species and loss of diversity resulted; Chlorophyceae and Dinophyceae were most affected. In contrast, macrophytic communities and the supported food chain remained affected for a much longer period. Fish utilizing a broad spectum diet showed no adverse effects, but organisms that depended on macrophytes for food and/or cover (tadpoles, bluegill, grass carp and benthic insect grazers) did poorly. Atrazine's half-life was on the order of several months; in absence of acquired resistance or tolerance, inhibition of photosynthesis could persist for some time. Adding grazing stress to macrophytes interacted with pesticide stress. Tadpoles were reduced due to loss of food, substrates for egg deposition, and loss of cover. Insects were reduced through loss of food (these were benthic grazers on periphyton and macrophyton).

Bluegill were reduced due to decreases in the number and kinds of insects that served as food organisms. In addition, loss of macrophytes reduced the refugia for bluegill young. Most of the observed population reductions were caused by indirect effects that would not be identified by laboratory exposures. Thus, field studies are important tools for determining potential effects of contaminants.

Stewart et al (1992) #254 investigated the effects of mcarophytes (*Potamogeton foliosus*) and filamentous algae contaminated with PCBs downstream of the settling basin in which they originated. The plants provided an energy source to the stream reach in which they settled. Snails (*Elimia* sp) grew more slowly on contaminated vegetation, leachates prepared from contaminated vegetations were toxic to C. dubia, and amphipods (*Gammarus* sp) preferred uncontaminated vegetation for forage. The authors concluded that energy subsidy and/or contaminants could explain the absence of mayflies and stoneflies in the stream reach downstream of the settling basin. Aquatic plants sequester contaminants and cycle them to overlying water. Such contaminants access to food web through herbivores or detritivores and may be physically released in different forms to the water column as the plants decompose. Zn, Cu, Hg, and Ni were also elevated in the plants to levels of concern.

Waara (1992) #213 examined the effects of Cu, Cd, and Pb and Zn on nitrate reduction in synthetic water and lake water. Fifty percent inhibition was found for a mixed heterotrophic culture of bacteria at 25, 85, 95 and 200 µg/L, respectively, in the synthetic medium. In lake water, approximately 25 percent reduction occurred at 10 µg/L Cu. Lower pH and phosphorous levels increased toxicity more than it was reduced by organic content. pH values were between 5.9 and 6.7 and total P between 10 and 22 µg/L.

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D-037255

APPENDIX A

D-037256

D-037257

Table . Summary of Toxicity Associated with Point – Source Dischargers

Discharger	Type	Flow (mgd)	Dilution	Acute Toxicity	Chronic Toxicity	Ambient Toxicity
East Bay Municipal Utility District	Municipal		1:10	18/18 – C. dubia 15/18 – M. beryllina	0/11 – C. dubia 1/18 – M. beryllina 4/11 – echino	not conducted
San Jose/Santa Clara	Municipal		1:10	2/19 – C. dubia 0/19 – M. beryllina	5/19 – C. dubia 1/19 – M. beryllina 2/15 – T. decipiens	4/8 – M. beryllina; South Bay
Chevron USA	Refinery		1:10	4/11 – M. beryllina	8/11 – M. beryllina 1/6 – echino 1/6 – bivalve	
Shell Oil Company	Refinery		1:10	14/17 – FHM 13/15 – M. bahia 8/16 – C. dubia	2/17 – FHM 3/15 – M. bahia 9/16 – C. dubia	
USS POSCO	Steel Mill		none	1/11 – C. dubia 2/12 – M. beryllina	6/12 – echino ?? – C. dubia ?? – M. beryllina	6/12 – echino 11/11 – C. dubia 2/12 – M. beryllina New York Slough
San Francisco: S.E. Water Pollution Control Plant	Municipal		1:10	not conducted	12/27 – bivalve	not conducted

D–037257

Gallagher and DiGiulio (1992) #177 investigated the role of fish gills in glutathione mediated detoxification of the fungicide chlorothalonil. Gills were found to be an important component in metabolism and detoxification of this chemical (the chemical undergoes direct conjugation with GSH without prior oxidation with cytochrome P-450). GSH levels increased within 72 hr of exposure and were maintained through the metabolism of cysteine. Davies (in #177) showed that chlorothalonil accumulates 1000 fold in rainbow trout gills and reduces lamellar diffusion. Species that have high capacity to detoxify elecropohilic contaminants may have an advantage in environments that contain such contaminants.

Kim et al. (1989) #99 compared the pathology of brown bullhead from contaminated and relatively clean sections of the Hudson River. The study section of river is contaminated with PCBs and, to a much lesser extent, with polychlorinated dibenzofurans and polychlorinated dibenzodioxins. The condition factor of fish from the contaminated site was lower than fish from the control site, but was not different between fish of different sexes from the same site. Muscle levels of PCBs in fish from the contaminated site averaged 38 μg/g, compared with 0.61 μg/g in fish from the control site. The sex ratio (M/F) was 0.25 at the contaminated site vs 1.2 at the control site. Incidence of gross abnormalities were similar in fish from both sites but histopathologic observations in spleen, kidney and liver were considered definitive markers of fish from the contaminated site. Bile duct hyperplasia was considered the most significant finding. Hemosiderin in liver, spleen, and kidneys suggested the breakdown of red blood cells; metals such as Cd, Pb and Hg were considered likely causative agents.

Baumann et al. (1991) #119 compared tumor frequencies in brown bullhead with concentrations of sediment contaminants in tributaries of the Great Lakes. Tumor frequency appeared related to polynuclear aromatic hydrocarbons (PAHs), rather than polychlorinated aromatic hydrocarbons. Hepatosomatic indices were also higher in association with PAHs. PAHs in the sediment from the contaminated site totaled 10.8 μg/g dry weight. Body burdens of PAHs in fish from contaminated site were 220 ng/g wet weight.

Pangrekar and Sikka (1992) #178 described the xenobiotic metabolizing enzymes in the liver

D-037258

Table 2. Application rates of carbofuran and methyl parathion and Sacramento River flows for the years 1970-1988.

	Application (lbs. X 1000)		
Year	Carbofuran	Methyl Parathion	River Flow (cfs X 1000)
1970	9	15	7.0
1971	9	30	14.9
1972	8	23	8.2
1973	11	32	8.9
1974	10	24	12.7
1975	5	23	15.0
1976	9	20	8.4
1977	21	17	5.7
1978	18	45	10.0
1979	29	67	6.7
1980	82	87	5.6
1981	108	100	6.9
1982	116	102	14.0
1983	73	54	23.1
1984	88	74	6.7
1985	58	48	5.4
1986	57	49	6.3
1987	57	57	6.9
1988	59	71	7.5

Table 1. Application rates of molinate and thiobencarb and Sacramento River flows for the years 1970-1985.

	Application (lbs. X 1000)		
Year	Molinate	Thiobencarb	River Flow (cfs X 1000)
1970	490	na	7.0
1971	797	na	14.9
1972	656	na	8.2
1973	601	na	8.9
1974	457	na	12.7
1975	962	na	15.0
1976	760	na	8.4
1977	598	na	5.7
1978	1300	na	10.0
1979	1400	na	6.7
1980	1600	8	5.6
1981	1700	287	6.9
1982	1500	675	14.0
1983	930	351	23.1
1984	1500	353	6.7
1985	1100	475	5.4

na = not applied.